UNDERSTANDING AND MITIGATING SEABIRD BYCATCH IN THE SOUTH AFRICAN PELAGIC LONGLINE FISHERY

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I dedicate this PhD to Barrie Rose, who was a champion of seabird conservation and research. Barrie's love of the ocean and all its wildlife was infectious; I have never met anybody as passionate about the ocean as Barrie. I learnt a tremendous amount from Barrie, who was always willing to impart his sea-faring wisdom, from the subtleties of seabird identification to the workings of the various fisheries. I will miss dinners with Barrie and Roselle, discussing our latest sea trips and pondering tricky seabird IDs. Barrie has had a great influence on many of the next generation of South Africa's birders and conservationists and will be sorely missed by all.

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ABSTRACT

Seabirds are considered one of the most threatened groups of birds in the world. They face additional mortality both on their breeding islands from introduced predators and at sea by fishing fleets, as fisheries bycatch, as well as other human impacts. Seabird bycatch has negatively affected many seabird populations worldwide, with trawl, gillnet and longline fisheries considered the most destructive to seabird populations. Seabird bycatch from trawl and gillnet fisheries has been significantly reduced in recent years, but large numbers of seabirds are still killed annually by longline fisheries. Of the two types of longline fisheries (demersal and pelagic), pelagic longlining is considered the most harmful to seabirds as lines remain closer to the surface for longer periods than demersal longlining, and it is harder to weight lines to ensure rapid sinking beyond the depth they are accessible to birds. Seabirds are killed when they swallow baited hooks and consequently drown. Despite the large number of studies investigating seabird bycatch from pelagic longline fisheries, there remain gaps in our understanding of seabird bycatch from pelagic longline fisheries. This thesis addresses some of these knowledge gaps and makes recommendations as to how seabird bycatch from pelagic longliners can be reduced at both a local and global scale.

Chapters 2 and 3 investigate the factors affecting seabird bycatch from pelagic longliners off South Africa, provide a summary of seabird bycatch from the fishery for the period 2006–2013 and quantify the structure of seabird assemblages associated with pelagic longline vessels off South Africa. This was achieved by analysing seabird bycatch data collected by fisheries observers as well as data from sea trials onboard pelagic longliners. Seabird bycatch by pelagic longliners off South Africa over the 8-year study period has been significantly reduced from the 8-year period (1998–2005), mainly driven by a significant reduction in seabird bycatch rates from foreign-flagged vessels, which are responsible for c. 80% of fishing effort off South Africa. Seabird bycatch rates from South African vessels still remain high, four times higher than the interim national target of < 0.05 birds per 1000 hooks. The species composition of seabird bycatch off South Africa is best explained by an understanding of the structure of the seabird assemblage associated with longline vessels. For most species, bycatch and attendance ratios were similar, but for some species such as shy-type and black-browed albatrosses there were large mismatches, likely caused by differences in foraging behaviour and foraging dominance hierarchies.

In Chapters 4 and 5 the foraging ecology of the most commonly recorded bycatch species off South Africa, the white-chinned petrel (*Procellaria aequinoctialis*), is investigated. An understanding of the foraging ecology of commonly recorded bycatch species enhances our understanding of seabird bycatch and helps to improve the design of current and future mitigation measures. The year-round movements of white-chinned petrels from Marion Island were investigated with Global Location Sensors (GLS loggers) and GPS loggers. Adult white-chinned petrels undertake only limited east-west movements of, with all birds remaining between southern Africa and Antarctica. These results strengthen the theory that there is limited spatial overlap year-round between white-chinned petrel populations from South Georgia, the southern Indian Ocean islands and New Zealand sub-Antarctic islands, suggesting that these populations can be managed as separate stocks. The diving behaviour of white-chinned and grey petrels (*P. cinerea*), another common bycatch species in Southern Ocean longline fleets, were examined with the use of temperature-depth recorders (TDRs), deployed on birds from Marion Island and Gough Island. My study was the first to analyse TDR dive data from any *Procellaria* petrel, and recorded them reaching maximum dive depths of 16 and 22 m, respectively. Current best practise suggests that baited hooks be protected to a depth of 5 m by bird-scaring lines, but my results suggest this depth should be increased to at least 10 m.

Although line weighting is a proven mitigation measure to reduce seabird bycatch from pelagic longliners, fishers have concerns that it will compromise fish catches, crew safety and operational efficiency. In Chapter 6 I analyse line weighting data from trials onboard three pelagic longline vessels, to address the concerns of fishermen. My results show that sliding leads can be incorporated into pelagic longline fisheries without compromising fish catch, crew safety or operational efficiency. I thus recommend that sliding leads be used on pelagic longline vessels fishing off South Africa.

By incorporating studies investigating the factors affecting seabird bycatch, seabird foraging ecology and the efficacy of seabird bycatch mitigation measures, my thesis has broadened our understating of seabird bycatch from pelagic longliners and makes meaningful recommendations to further reduce bycatch, both locally and globally. Although seabird bycatch rates have declined off South Africa, through the use of a number different mitigation measures, they still remain higher than the South African national target and thus more work is needed to achieve this target. To reduce seabird bycatch from pelagic longliners to acceptable levels, studies from the world's various longline fleets needs to be considered and improved upon, with seabird conservationists and fishermen working together to achieve this goal.

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CONTENTS

PAPERS ARRISING FROM THESIS

The following manuscripts were published prior to submission of this thesis. These manuscripts have multiple authors, each of which contributed in some way to its production. In all cases D.P.R. conducted the analyses, while D.B.J and D.D deployed and retrieved devices on the breeding islands. All authors contributed to the writing of the paper, which was led by D.P.R. Contributions: Rollinson 75%, Dilley (and Davies) 5% and Ryan 20%

Rollinson DP, Dilley BJ, Ryan PG. (2014). Diving behaviour of white-chinned petrels and its relevance for mitigating longline bycatch. *Polar Biology* 37: 1301-1308.

Rollinson DP, Dilley BJ, Davies D, Ryan PG. (2016). Diving behaviour of grey petrels and its relevance for mitigating long-line by-catch. *Emu* 116 DOI: 10.1071/MU15032.

The following manuscript has been accepted for publication, but has not yet been published. D.P.R. undertook all analysis, and the writing of the manuscript, while P.G.R. conducted all post-mortem investigations. Fisheries observer data were provided by the South African Department of Agriculture, Forestry and Fisheries. R.M.W and P.G.R. acted in a supervisory capacity, aided in the study design and oversaw the writing of the manuscript.

Rollinson DP, Wanless RM, Ryan PG. (in press). Patterns and trends in seabird bycatch associated with the pelagic longline fishery off South Africa. *African Journal of Marine Science*.

LIST OF ABBREVIATIONS AND ACRONYMS

ACAP: Agreement on the Conservation of Albatrosses and Petrels

AIC: Akaike's information criterion

ATF: Albatross Task Force

CCAMLR: Convention for the Conservation of Antarctic Marine Living Resources

CCSBT: CCSBT Commission for the Conservation of Southern Bluefin Tuna

EEZ: Exclusive Economic Zone

FAO: Food and Agriculture Organisation of the United Nations

GLS: Global Location Sensor

GPS: Global Positioning System

IBA: Important Bird Area

ICCAT: International Commission for the Conservation of Atlantic Tunas

IOTC: Indian Ocean Tuna Commission

NPOA: National Plan of Action

PEI: Price Edwards Islands

RFMO: Regional Fisheries Management Organisation

SST: Sea surface temperature

TDR: Temperature-depth recorder

CHAPTER 1

INTRODUCTION TO SEABIRD BYCATCH BY PELAGIC LONGLINE FISHERIES

1.1.1 Seabirds at risk

Seabirds are considered one of the most threatened groups of birds in the world, with pelagic species substantially more threatened than coastal species (Croxall et al. 2012). Of the seabird orders, penguins (Sphenisciformes) and albatrosses/petrels (Procellariiformes) are the most threatened (Croxall et al. 2012). Many procellariiform species are particularly susceptible to increased mortality as they are long-lived, late to mature, have low reproductive output and rely on K-selected life-history patterns (Lewison et al. 2004). On their breeding islands one of the main threats facing procellariiform species is the introduction of alien predators. Having evolved to breed in areas largely lacking mammalian predators, Procellariiformes are unable to defend themselves against introduced mammals including cats (*Felis catus*), rats (*Rattus* spp.) and mice (*Mus* spp.; Courchamp et al. 2003). Introduced mammalian predators are thought to be responsible for the extirpation and decline of many seabird species worldwide (Jouventin et al. 2003).

Within the Procellariiformes, many species are extremely wide-ranging (to the extent of regularly circumnavigating the planet), especially as non-breeders or juveniles (Weimerskirch et al. 2006), frequently traveling through multiple national Exclusive Economic Zones (EEZs) as well as areas beyond national jurisdiction. The wide ranging habits of many Procellariiformes mean that, despite breeding within the EEZ of one country, they may be at risk whilst foraging within the EEZs of many other countries, and in waters beyond national jurisdiction. Seabirds are impacted by commercial fisheries by changing the availability of food sources, either through direct competition for food or indirectly by altering food web structures, and mortality as non-target catch or bycatch (Furness 2003). Seabird mortality from fisheries bycatch is considered of similar significance as the effects of alien invasive species, driving the decline of seabird populations worldwide, although this varies between species (Croxall et al. 2012).

1.1.2 Global seabird bycatch from commercial fisheries

Fisheries represent one of the key threats to marine biodiversity, impacting marine megafauna from a wide range of taxa; from the coast to the open ocean and from the tropics to the poles (Halpern et al. 2008). Marine ecosystems are primarily damaged by commercial fisheries in three ways; over-harvesting of marine resources, habitat degradation, and mortality of non-target species/bycatch (Halpern et al. 2008). Fisheries bycatch not only kills seabirds but is seen as one of the primary causes in observed declines of marine mammals and sea turtles (Lewison et al. 2014). Different

fisheries kill seabirds in very different ways, with some of the most damaging including gillnet, trawl and longline fisheries (Bull 2007; Lewison et al. 2014).

Significant mortality of seabirds has been identified in a number of trawl fisheries from around the world (Weimerskirch et al. 2000; Sullivan et al. 2006a; Watkins et al. 2008; Abraham 2010). Most birds are killed when colliding with cables or by becoming entangled in the net while attempting to feed (Sullivan et al. 2006a, b; Melvin et al. 2010). Most birds are killed when their wings get wrapped around moving cables, whilst flying or in the water, which results in their being pulled underwater and drowning (Watkins et al. 2008; Melvin *et al*. 2010). Some of the most effective mitigation methods for reducing seabird bycatch from trawl fisheries include; offal and discard management, the banning of net monitoring/net sonde cables, the use of paired bird-scaring lines, and reducing the time the net is on or near the sea surface (Bull 2007). Through the use of bird-scaring lines, Maree et al. (2014) measured a 73–95% decrease in seabird mortality from demersal trawl vessels operating off South Africa. Similar reductions in seabird bycatch from trawl fisheries have been noted from the Falkland Islands (Sullivan et al. 2006a) and the eastern Bering Sea (Melvin et al. 2010).

Gillnet fisheries have caused substantial reductions in seabird numbers, mostly affecting deep-diving species of alcids, which are confined to the Northern Hemisphere (Melvin et al. 1999), although penguins also have been affected in the Southern Hemisphere (Ryan and Cooper 1991). Seabirds become entangled in gillnets whilst diving for fish or benthic prey. Seabird mortality from gillnets depends on a number of factors such mesh size, setting depth, time of day and other factors (Žydelis et al. 2013). The huge numbers of birds killed each year in the North Atlantic (Tull et al. 1972) and North Pacific Oceans (DeGange et al. 1993) was a contributing factor towards the global ban on the use of drifting gillnets in international waters, imposed in 1990 by the United Nations (Alverson et al. 1994). A number of bycatch mitigation techniques have been successfully trialled in gillnet fisheries, some of these include; spatio-temporal closures, visual alerts on fishing gear, acoustic alerts attached to the nets, and increasing the setting depth of nets (Žydelis et al. 2013). Despite these mitigation measures, seabird bycatch remains high in many of the temperate and sub-polar regions of the world (Žydelis et al. 2013). The 1990 ban on drifting gillnets in international waters meant a large increase in the number of high seas longline fisheries targeting large pelagic fishes such as tuna (Scombridae) and billfishes (Istiophoridae; Melvin and Parrish 2001).

Bycatch from longline fishing has had devastating effects on many species of marine megafauna throughout the world's oceans (Lewison et al. 2004). Sea turtles, sharks and seabirds have been worst affected, with substantial population declines of all three groups, attributable to longline

fishing activities (Lewison et al. 2004). Seabird bycatch from longlining is almost exclusively confined to surface-scavenging species and thus species such as penguins, alcids and cormorants are rarely caught on longlines (Brothers et al. 1999). The configuration of the longline, type and number of hooks and branchline length all vary between fisheries, depending on the target species (Gilman 2001). Longline fishing can be broadly classified as either targeting demersal or pelagic fish species. Demersal longlines set lines along or close to the seafloor with anchors/weights at either end of the mainline (Melvin et al. 2001). Demersal longliners can set more than 10 000 hooks per day, typically using 1-m long branchlines (Gilman 2001). Demersal longline bycatch predominantly occurs in fisheries targeting Patagonian toothfish (*Dissostichus eleginoides*) or hake (*Merluccius* spp.) in the Southern Ocean (Gilman 2001). Hake longline fisheries off southern Africa initially caught birds at high rates (Barnes et al. 1997), however seabird bycatch from this fishery has been greatly reduced through the use of appropriate mitigation measures (Petersen et al 2009a). Pelagic longline vessels typically fish close to the sea surface and set longlines which are regularly over 100 km long, with up to 3500 hooks attached, using much longer branchlines than demersal longlines (Gilman 2001). Pelagic longline bycatch occurs in fleets targeting tuna and billfish species throughout the world's oceans (FAO 1997; 1998; Yeh 2013). Seabird bycatch from demersal longline fishing is easier to mitigate for as line sink rates are generally faster than pelagic longlines (Melvin et al. 2013). As demersal longlines use much shorter branchlines it means all fishing gear generally sinks below the surface within 50 m of the vessel, while for pelagic longlines which use much longer branchlines, hooks sink slowly and independent of the mainline and are thus available to scavenging seabirds for longer periods of time (Melvin et al. 2013).

Seabird bycatch from both pelagic and demersal longliners is considered a significant contributor to extinction risks for many species (see Anderson et al. 2011 for review). Encouragingly, seabird mortality from many demersal longline fisheries has seen reductions in recent years, mostly attributable to the use of a number of mitigation measures (Croxall and Nicol 2004; Melvin et al. 2001; Petersen et al. 2009a). Mitigation measures proven to reduce seabird bycatch include; underwater setting funnels, bird-scaring lines, integrating weight into the fishing line and night-time setting, although these measures work best when used simultaneously (Brothers et al. 1999). Underwater setting funnels ensure that baited hooks are unavailable to scavenging seabirds until outside the funnel, usually at a depth of 1–2 m (Brothers et al. 1999; Ryan and Watkins 2002). Bird-scaring lines are composed of highly visible streamers attached to a mainline which are then flown behind the vessel during the setting process. The streamers are blown around by the wind and their erratic movement largely prevents birds from landing in the area covered by the bird-scaring line. Bird-scaring lines reduce seabird bycatch in demersal longline fisheries by ensuring seabirds avoid

the area immediately behind the vessel where baited hooks are closest to the surface (Dietrich et al. 2008). By integrating weight into demersal longlines, the lines sink quicker, which means less time is available for scavenging seabirds to retrieve baited hooks (Dietrich et al. 2008). If longlines are set at night, seabird bycatch is likely to be reduced as fewer seabird species forage at night than during the day and for those species which are able to forage nocturnally, reduced visibility ensures they forage less efficiently (Brothers et al. 1999). As seabird bycatch from demersal longliners can be mitigated more easily than seabird bycatch from pelagic longlining (Melvin et al. 2013), seabird bycatch from pelagic longlining could be considered the 'last challenge' for mitigating seabird bycatch from longlines.

My thesis thus aims to provide an in-depth understanding of the magnitude of seabird bycatch from pelagic longline vessels off South Africa as well as the factors affecting seabird bycatch. I analyse seabird bycatch data spanning eight years (2006–2013), during which time a number of changes to permit conditions were introduced. As a result, I am able to link changes in bycatch levels to changes in mitigation measures over time. With an understanding of all possible seabird bycatch mitigation measures tested globally, together with a thorough understanding of the workings of the South African pelagic longline fishery, I make recommendations as to which mitigation measures might be applicable locally and those which could be appropriate for other pelagic longline fisheries worldwide.

1.1.3 Seabird bycatch from pelagic longline fisheries

Seabird bycatch from pelagic longline fisheries was first documented in the late 1980s by Japanese tuna fleets operating off Australia and New Zealand (Brothers et al. 1991; Murray et al. 1993). Seabirds caught on longlines typically fall into two feeding strategies; those (generally larger) species such as albatrosses and giant petrels (*Macronectes* spp.), which feed on the surface or undertake shallow dives, and those (generally smaller) species, such as petrels and shearwaters, which undertake deep dives. The deeper-diving species have been implicated in the bycatch of larger surface-scavenging species as they return baited hooks to the surface which would otherwise have been unavailable to the larger, surface-scavenging species (Jiménez et al. 2012).

Most birds are caught during line setting operations when the baited hooks are within reach of diving seabirds (Brothers et al. 2010); scavenging seabirds swallow baited hooks, become hooked and drown, with their bodies only recovered during hauling. It is important to note that many of the seabirds which are killed by longlines during line setting may not be recovered during hauling

operations. Brothers et al. (2010) estimated that nearly half the birds observed caught during setting were not recovered during hauling, having been lost due to scavenging, currents or other mechanical action during the line soak (Brothers et al. 2010). A smaller number of seabirds become hooked during hauling operations, when hooks are returned to the surface and again made available to scavenging seabirds. These birds are less frequently killed as fishers are able to dehook the birds before they have time to drown (Gilman 2011), but the severity of injuries sustained while hooked is largely unknown (Brothers et al. 1999). Seabird mortality is also thought to occur as a result of hooks remaining in released birds, or hooks ingested within discarded offal or bycatch (Brothers et al. 1999). However seabirds are able to digest hooks (but not plastic snoods), and albatrosses have been observed breeding with hooks protruding from their necks (PG Ryan pers. comm.), suggesting that they can tolerate seemingly severe injury, however these are unlikely to be birds which were brought aboard fishing vessels for dehooking; birds brought aboard for dehooking may sustain more serious injuries.

Information on seabird bycatch from commercial fishing vessels is predominantly determined by the placement of fisheries observers onboard these vessels. However observer protocols for data collection are not standardised across global fisheries and are mostly geared towards the collection of fish and fisheries data, rather than seabird interaction data (Gilman et al. 2005). The accuracy of data collected by fisheries observers also depends on the experience and commitment of the observer. For example, observers can seldom monitor an entire haul and because of the deliberate or accidental dislodgement of seabird carcasses from fishing hooks (Gilman et al. 2005), observers are likely to regularly under-record seabird bycatch. Therefore, as suggested by Brothers et al. (2010), seabird bycatch estimates, based on haul data alone, are likely to significantly under-record actual bycatch levels.

1.1.4 Mitigation measures to reduce seabird bycatch from pelagic longline fisheries

A suite of mitigation measures and devices has been demonstrated to reduce seabird bycatch in pelagic longline fisheries. However, for a mitigation measure to be successfully incorporated into a fishery's standard operating procedure it should ideally ensure that seabird bycatch is reduced without negatively affecting fish catches, operational efficiency or fisher safety (Gilman 2011). Most mitigation measures are designed to reduce seabird bycatch during setting operations, but some also aim to reduce seabird bycatch during hauling, such as offal management and bird curtains (Gilman et al. 2014). Certain seabird bycatch mitigation measures may not necessarily work in all

fisheries and so the relevant suite of measures needs to be designed for individual fisheries (Gilman 2011). Most seabird bycatch mitigation measures in pelagic longline fisheries work best when they are used simultaneously with other mitigation measures; the simultaneous use of bird-scaring lines, line weighting and night-time setting is considered 'best practise' to reduce seabird bycatch (Melvin et al. 2014; ACAP 2016). As suggested by Løkkeborg (2008), mitigation measures designed for setting operations can be placed into four broad categories:

1. Those which avoid fishing at times when seabird interactions are more likely (e.g. night-time setting or time-area closures, such as around breeding colonies during the breeding season).
2. Those that limit access to baited hooks (weighted branchlines, thawed bait, bait-casting machines, underwater setting devices, hook-shielding devices).
3. Those that deter birds from taking baited hooks (bird-scaring lines).
4. Those that reduce the attractiveness or visibility of baited hooks (offal management, bait type, dyed bait).

By setting longlines at night, peak periods of seabird foraging activity are avoided by longline vessels. Many seabirds, particularly albatrosses, forage less frequently during the night as the reduced light levels result in less productive foraging opportunities (Phalan et al. 2007). Seabird bycatch rates on longline fisheries are consistently higher when lines are set during daylight hours, compared to night-time setting (Cherel et al. 1996; Petersen et al. 2009b; Melvin et al. 2013, but see Barnes et al. 1997). Despite this, night-time setting is less effective at reducing the bycatch of petrel species (Murray et al. 1993), which regularly forage nocturnally, when there is reduced competition with other seabird species(Mackley et al. 2011). For night-time setting to be most effective setting should be completed at least two hours before nautical dawn so as to not extend into daylight hours (Murray et al. 1993; Barnes et al. 1997). Many seabird species increase their foraging activity during periods of high lunar luminance (Phalan et al. 2007), and thus, particularly on the days around full moon, it is important that night setting is used in conjunction with other mitigation measures to reduce the bycatch of nocturnally active foragers, such as petrels (Melvin et al. 2013).

The addition of weights to branchlines is considered one of the most important measures for reducing seabird bycatch. Line weighting increases the sink rate of baited hooks, carrying them below the depth accessible to seabirds more rapidly, ideally while the line is still protected in part by one or more bird-scaring lines (Robertson et al. 2010). Fishers are, however, hesitant to add weight to branchlines because of the concern that it could negatively affect fish catches (Brothers et al. 1999), and due to safety risks associated with branchline weighting (Sullivan et al. 2012). Despite the importance of line weighting as a seabird mitigation measure, there are few published studies on the

relationship between gear configuration and baited hook sink rates, or how line weighting affects fish catch and fishing operations. Robertson et al. (2010) found that adding weights to branchlines with long leaders (distance between hook and weight) had little effect on hook sink rates and thus is unlikely to significantly reduce seabird bycatch rates. Melvin et al. (2013) tested a number of different weighting options within 1 m of the hook and concluded that seabird bycatch was significantly reduced on branchlines weighted with at least 60 g, without any measurable effect on target fish catch. For line weighting to be accepted by fishers, it is vital that more studies assess the effect of line weighting on target fish catches, crew safety and operational efficiency.

Frozen bait sinks more slowly than thawed bait, and thus it is recommended that all bait used during setting be sufficiently thawed to increase hook sink rates (Brothers et al. 1999). However, fishers often prefer to use partly frozen bait, because thawed fish is more likely to be pulled off the hook while being cast, either manually or automatically (Brothers et al. 1999).

Bait-casting machines ensure that bait is consistently landed in a similar area while being cast, and thus baited hooks are avoided from landing in turbulent water (created by propellers), where they tend to sink slower than lines cast into calmer water, potentially increasing seabird bycatch rates (Brothers et al. 1999). Crew often prefer to use bait-casting machines as it is easier than manually casting thousands of baited hooks (pers. obs.). By ensuring lines consistently land in a similar area, bait-casting machines also ensure that baited hooks land in an area under the protection of bird-scaring lines.

Bird-scaring lines in pelagic longline fisheries work in much the same way as those used by demersal longline vessels (as described above); by ensuring that seabirds are deterred from a fixed area behind the vessel where hooks are close to the surface. Bird-scaring lines are more appropriate for reducing seabird bycatch from demersal longlines than pelagic longlines (Løkkeborg 1998); the slower sinking pelagic longlines mean that bird-scaring lines need to cover a much larger distance behind the vessel than demersal longlines (Løkkeborg 1998). The use of paired, rather than single bird-scaring lines during pelagic longlining has shown to reduce seabird attacks on baited hooks significantly, consequently reducing seabird bycatch (Sato et al. 2013). The effectiveness of bird-scaring lines is however much dependent on line sink rates achieved (by applying weights to branchlines); the distance a bird-scaring line is required to protect baited hooks is dependent on the distance lines sink beyond the diving capabilities of foraging seabirds (Melvin et al. 2014).

By reducing the attractiveness of longline vessels to seabirds, the size of the seabird assemblages associated with these vessels is likely to be reduced (Brothers et al. 1999). One way to achieve this is

to ensure offal and unused bait are not discarded during setting or hauling, but rather discarded once all fishing activities have ceased. The visibility of baited hooks can be reduced by dyeing bait to match the water colour, which can result in the reduced foraging efficiency of the seabirds. However, the effectiveness of dyed bait at reducing seabird bycatch has varied considerably between studies (Gilman et al. 2016).

By encasing the point and barb of a hook, hook-shielding devices ensure that no seabirds can be hooked until the hook-shielding device releases the hook, at a prescribed depth or length of time after setting (Wolfaardt et al. 2016). Two such devices for which initial trials have been conducted include the hook pod (Sullivan et al. 2016) and Smart Tuna Hook (GB Baker unpubl. data). The hook pod encapsulates the hook and barb and is pressure activated to open at a prescribed depth, releasing the hook and allowing fishing to commence (Sullivan et al. 2016). The hook pod is attached to the branchline and thus can be reused once retrieved during line hauling. Smart Tuna Hooks have steel caps placed over the baited hook, covering the barb, which are held in place by a pin that corrodes in saltwater after roughly 15 min, allowing fishing to commence. The cap is not attached to the line and thus sinks once released; it is thought to corrode fully within 12 months. More research into the feasibility of introducing hook-shielding devices into pelagic longline fisheries still needs to be conducted; the precise design of the devices mean that they may be feasible for some fishing fleets and not for others.

1.1.5 International seabird bycatch governance

Internationally, fisheries are bound by law to protect and preserve the marine environment, and to consider the effects of fishing on species associated with, or dependent upon commercially exploited species (1982 UN Convention of the Law of the Sea, UNCLOS). This is further elaborated by the 1995 Fish Stocks Agreement, which compels signatory states to minimise bycatch and impacts on associated and dependent species. Additionally, pelagic longline fleets are also governed by Regional Fisheries Management Organisations (RFMOs), intergovernmental organisations and fishing nations (Nel and Taylor 2003).

Globally, 13 RFMOs have been established to manage high seas fisheries, of which five are specifically for tunas and tuna-like species: the Commission for the Conservation of Southern Bluefin Tuna (CCSBT), Indian Ocean Tuna Commission (IOTC), International Commission for the Conservation of Atlantic Tunas (ICCAT), Inter-American Tropical Tuna Commission and Western and Central Pacific Fisheries Commission (Gilman 2011). Signatory nations to RFMOs must ensure all vessels operating

under their flag abide by the relevant resolutions (including seabird bycatch mitigation measure compliance) while fishing in international waters.

The Food and Agriculture Organisation (FAO) is one of the most important intergovernmental organisations governing marine fishing operations and hence seabird bycatch. The FAO's International Plan of Action for Reducing Seabird Bycatch (IPOA-Seabirds; FAO 1999) is non-binding and aims to give states general guidelines on bycatch regulations (Trouwborst 2008). The IPOA aims to reduce incidental bycatch of seabirds in longline fisheries by setting out activities which should be performed by member states. These include performing assessments to determine whether incidental bycatch is indeed a problem by vessels fishing within the state's maritime zone, or by vessels flying the state's flag outside these zones (Trouwborst 2008). Countries also are encouraged to draft a National Plan of Action (NPOA) to address seabird bycatch (Trouwborst 2008). If a seabird bycatch problem does occur in the country then further action can be taken, however unless the NPOA has been included in national legislation, it may not mandatory for action to be taken. A number of states have set thresholds in their NPOA to reduce bycatch to an interim target of < 0.05 birds per 1 000 hooks (Tuck 2011), including South Africa, which adopted its NPOA in 2008 (Cooper et al. 2008). South Africa is also a member of the Agreement on the Conservation of Albatrosses and Petrels (ACAP), which is a binding agreement to develop resolutions and advice to reduce seabird bycatch (ACAP 2015).

1.1.6 Seabird bycatch from pelagic longline fisheries off South Africa

The earliest record of longline fishing off South Africa dates back to the 1950s when Japanese vessels began targeting tuna species; mostly albacore (*Thunnus alalunga*), southern bluefin *(T. maccoyii*) and bigeye (*T. obesus*; Ryan et al. 2002). This fleet was soon followed by Taiwanese and domestic South African vessels in the 1960s, both targeting tunas (Penney and Griffiths 1999). Fishing effort was reduced in the domestic fleet in the mid-1960s and thereafter mostly conducted by Japanese and Taiwanese vessels under bilateral agreements. In 1997, 30 experimental permits were issued to South African-flagged vessels targeting swordfish (*Xiphias gladius*) and tuna under new bilateral agreements with Japan and Taiwan (Ryan et al. 2002). Initially the South African fleet targeted bigeye tuna, but substantial catches of swordfish caused many vessels to target this species instead. By 2000, swordfish catches had dropped significantly and fishery catch was dominated by yellowfin (*T. albacares*) and bigeye tunas (Ryan et al. 2002). Fifty permits were issued in 2004; 30 tuna-

directed and 20 swordfish-directed. The number of permits was reduced in 2005, with 26 tuna-directed and 17 swordfish-directed permits issued.

The most recent study summarising seabird bycatch from pelagic longline fishing off South Africa was undertaken by Petersen et al. (2009b), which detailed seabird bycatch by Asian and local longline vessels for the period 1998–2005. By extrapolation they estimated as many as 2900 birds were killed per year, however a decreasing trend was noted with approximately 5 900 birds killed in 1998 and 1 800 birds killed in 2005. At that stage the pelagic longline fishery off South Africa was comprised of the local South African fleet (~1.3 million hooks per year) and the Asian joint venture fleet (~5.2 million hooks per year) targeting swordfish and tuna, respectively (Petersen et al. 2009a). Petersen et al. (2009b) found large differences in bycatch rates between Asian and South African vessels; Asian vessels caught 88% of seabird bycatch at a rate of 0.51 birds per 1000 hooks, while South African vessels caught 12% at a rate of 0.23 birds per 1000 hooks. Many variables affected seabird bycatch rates, including the use of mitigation devices, vessel flag, moon phase, season, area, time of set, etc., but the most important factor was individual vessel. Although perhaps surprising, this vessel effect occurs because skipper behaviour and fishing operations differ between individual vessels, to the extent that 20% of vessels killed 74% of seabirds, with one vessel catching 23% of all seabirds, at a rate of 2.3 birds per 1 000 hooks (Petersen et al. 2009b).

Petersen et al. (2009b) found that seabird bycatch in the pelagic longline fishery off South Africa was comprised mostly of white-chinned petrels (*Procellaria aequinoctialis*; 69%), with smaller numbers of shy-type (*Thalassarche cauta/steadi*; 24%), black-browed (*T. melanophris*; 4%) and Indian/Atlantic yellow-nosed albatross sp. (*T. carteri/chlororhynchos*; 2%); the remaining 1% of bycatch comprised an additional 6 species. Of the most commonly recorded species, two are currently regarded as endangered (Indian and Atlantic yellow-nosed albatross), one vulnerable (white-chinned petrel) and two near-threatened (shy-type and black-browed albatross; IUCN 2016).

1.2 Overview of the thesis

Despite seabird bycatch from pelagic longliners having been identified almost three decades ago, there are still large knowledge gaps in our understanding of the complex problem. To ensure a complete understanding of seabird bycatch from the fishery, it is vital to gain knowledge of the fishery and fishing operations as well as knowledge of the ecology of bycatch-prone seabird species. Although there are many similarities among the various longline fleets operating throughout the world's oceans, there are also specific differences, depending on a range of factors. As a result, some

recommendations to reduce seabird bycatch may be applicable to longline fleets worldwide, while others may only be applicable to certain fleets. Despite much of my research being focussed on seabird bycatch off southern Africa, many of my findings are relevant for pelagic longline fishing fleets worldwide. Chapters 2 and 3, although focussing on factors influencing seabird bycatch off southern Africa, provide advice and recommendations which will be useful to understand and reduce seabird bycatch in other fleets worldwide. The research conducted on the foraging ecology of white-chinned and grey petrels (*P. cinerea*; Chapters 4 and 5) is relevant to all other fisheries where the bycatch of these two species is considered a problem; in both demersal and pelagic longline fleets. My line weighting study (Chapter 6) may be specific to Korean-style pelagic longliners, however it does highlight the need for future seabird mitigation trials to focus equally on fishers' concerns as much as the potential of the measure to reduce seabird bycatch.

1.2.1 Chapter descriptions

Seabird bycatch in the South African pelagic longline fishery has previously been summarised in two studies between 1998 and 2005 (Ryan et al. 2002; Petersen et al. 2009b). Chapter 2 updates these studies by summarising seabird bycatch in the South Africa pelagic longline fishery for the period 2006–2013. Importantly, it also re-assesses the various factors influencing seabird bycatch in the region, and considers why there has been a decrease in catch rates compared to the late 1990s and early 2000s. This study makes use of fisheries observer data (various fishery related information) as well as autopsied seabird data (confirmation of species, age and sex).

Chapter 3 investigates the relationship between the assemblage of seabirds attending pelagic longline vessels and seabird bycatch off southern Africa. Daily seabird counts were conducted in three different fishing regions throughout southern Africa, to compare seabird bycatch with the community of birds attending longline vessels. This is the first study to document the structure of the seabird assemblage at longline vessels off southern Africa, providing interesting insights into the makeup of seabird bycatch within the subregion.

Chapter 4 investigates the year-round movements of white-chinned petrels; the most commonly recorded bycatch species in southern oceans' pelagic and demersal longline fisheries. By deploying Global Location Sensors (GLS loggers) and Global Positioning System (GPS) loggers on white-chinned petrels breeding on Marion Island, southwest Indian Ocean, I identify the degree of overlap of foraging distribution with longline fisheries, as well as core foraging areas. Although studies on the

movements of white-chinned petrels from other breeding islands have been undertaken, this is the first such study of birds from the Prince Edward Island (PEI) population.

A species' diving capability is a useful indicator of its likely vulnerability to mortality from longline fisheries, with deeper-diving species having better access to baited hooks during line setting (Jiménez et al. 2012). Critically, knowledge of diving ability, combined with hook sink rates and the aerial extent of protection afforded by bird-scaring lines, determine the probability of a seabird bycatch event occurring. Chapter 5 summarises the results of the first deployment of temperature-depth recorder (TDR) loggers on white-chinned and grey petrels. An understanding of the diving ecology of commonly recorded bycatch species helps to provide better guidelines as to what depth longline hooks need to be protected to during line setting.

Chapter 6 investigates the potential of sliding leads as a new line weighting technique which can be used to mitigate seabird bycatch on pelagic longline vessels. Sliding leads were trialled onboard three Korean longline vessels operating off southern Africa and western Australia. This study focusses primarily on the concerns fishers have raised with regards to adding weight to branchlines, how it will affect fish catches, operational efficiency and crew safety.

Chapter 7 is a synthesis of my research. Here I make recommendations on how seabird bycatch can be tackled both locally and on a global scale. I also highlight the insights gained from many months onboard both domestic (South African) and foreign (Japanese and Korean) longline vessels as a fisheries observer and researcher. I suggest how these insights can be useful for organisations and legislation governing seabird bycatch from pelagic longline fisheries.

1.2.2 Author contributions

Apart from some of the field work, lab work and assistance with data analysis, this work is all my own. My supervisor Peter Ryan performed all the autopsies on seabirds, while fisheries observers obtained the fisheries data onboard pelagic longline vessels; I make use of both of these databases in Chapter 2. Ben Dilley and Delia Davies deployed and retrieved the GLS devices on white-chinned petrels on Marion Island (Chapter 4), and also deployed and retrieved the GPS devices on white-chinned and grey petrels on Marion and Gough islands, respectively (Chapter 5). In chapter 6, data from the two research trips trialling sliding leads off western Australia were collected by Gi Chul Choi, while I was assisted by Park Jeong-Yun off South Africa/Mozambique. Chevonne Reynolds and Dominic Henry advised me on data analysis throughout my thesis. Jaimie Cleeland allowed me to use

her modified R package to analyse GLS data (Chapter 4) while Otto Whitehead provided advice on data analysis for both the GLS (Chapter 4) and TDR data (Chapter 5).

1.2.3 Chapter structure

My thesis is written as a series of stand-alone chapters to facilitate their publication. The structure of each chapter follows that of a scientific paper and includes an abstract, introduction, methods, results and discussion. As such, there is some repetition, particularly within the chapter introductions, although I have tried to keep this to a minimum. Some of the chapters have already been accepted for publication in peer-reviewed journals; the versions that appear here have been edited and formatted to fit into the thesis format.

CHAPTER 2

PATTERNS AND TRENDS IN SEABIRD BYCATCH ASSOCIATED WITH THE PELAGIC LONGLINE FISHERY OFF SOUTH AFRICA

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Abstract

Both foreign and domestic pelagic longline vessels operate in the South African EEZ and adjacent international waters where they kill hundreds of seabirds each year as bycatch. In order to update the assessment of the impact of the pelagic longline fishery on seabirds off South Africa, information on autopsied seabirds and national fisheries observer bycatch records were summarised for 2006–2013. Foreign-flagged (Asian) vessels had 100% observer coverage throughout the study period, whereas only 6% of fishing effort by South African-flagged vessels was observed (with no coverage in 2011–2013). Vessels with observers caught seabirds at a rate of 0.132 birds per 1000 hooks, resulting in the estimated mortality of 2 851 individuals (361 per year) comprising 14 species. Extrapolation of observed fishing sets to unobserved fishing sets by the South African domestic longline fleet suggested that c. 750 additional birds were likely killed during the study period; therefore a combined total of c. 450 birds were killed per year. White-chinned petrels were most frequently killed (66%), followed by shy-type albatrosses (21%), black-browed albatrosses (7%), Indian yellow-nosed albatrosses (3%) and Cape gannets (2%). Bycatch rates were 70% lower than in 1998–2005. Vessel nationality, time of setting, moon phase, year, season, area and bycatch mitigation measures all influenced seabird mortality. Concurrent with 100% observer coverage, significant reductions in bycatch rates occurred in the Asian fleet in the latter years of the study, and these rates now approximate the national target (0.05 birds per 1000 hooks). However, bycatch rates remain high in the South African fleet, where no observers were deployed during 2011–2013, highlighting the need for independent observer programmes in fisheries – a matter of global interest. Suggestions are made as to how seabird bycatch by pelagic fisheries off South Africa may be further reduced.

2.1 Introduction

Fisheries bycatch is a factor causing significant population decreases for many seabird populations globally, particularly among albatrosses and the larger petrels (Croxall et al. 2012), with longline fishing considered one of the most destructive types of commercial fishing affecting seabirds (Croxall 1998). Pelagic longline fleets are active throughout the world's oceans (Anderson et al. 2011). However, seabird bycatch is of greatest concern in longline fleets of the Southern Hemisphere, where large numbers of threatened seabirds forage and are regularly recorded as bycatch (Nel and Taylor 2003).

For several decades the productive waters off South Africa have supported fleets of pelagic longline vessels primarily targeting tunas and swordfish as well as blue (*Prionace glauca*) and mako sharks (*Isurus oxyrinchus*; Petersen et al. 2009a). These waters also support large numbers of foraging seabirds (Crawford et al. 1991). Eleven species of seabird have previously been recorded as bycatch in pelagic longline fleets operating off South Africa (Petersen et al. 2009a), including one species listed by the IUCN as critically endangered, two as endangered and two as vulnerable (IUCN 2016).

Historically, pelagic longline fishing killed far greater numbers of seabirds off southern Africa than demersal longline fishing (Petersen et al. 2009a, 2009b). Between 1998 and 2000, the seabird bycatch rate for the pelagic longline fleet operating in the South African EEZ and adjacent international waters was 1.60 birds per 1 000 hooks, killing an extrapolated estimate of between 19 000 and 30 000 birds per year (Ryan et al. 2002). Between 1998 and 2005, the bycatch rate decreased to 0.44 birds per 1 000 hooks or 2 890 birds per year (Petersen et al 2009a). Both these estimates are well above the South African NPOA target for reducing seabird bycatch, which aims to limit seabird bycatch to below a rate of 0.05 birds per 1 000 hooks for vessels fishing within South African waters (DEAT 2008). This paper reports bycatch rates for the period 2006–2013.

Seabird bycatch mitigation measures within South African fishing permit conditions have been slightly amended throughout the study period (Table 2.1); however, some of the most important mitigation measures include: foreign-flagged vessels fishing under joint venture agreements within the South African EEZ must ensure lines are set between nautical sunset and nautical sunrise, with a bird-scaring line deployed during setting operations. When fishing under joint venture agreements on the high seas, foreign-flagged vessels are permitted to set lines during the daytime as long as a minimum of 60 g of weight is attached to branchlines within 2 m of the hook (DAFF 2013). Permit regulations are similar for vessels targeting swordfish (South African-flagged). However, fishers need to ensure that a minimum of 60 g of weight is added to branchlines within 2 m of the hook at all times. Given this additional weight, these vessels are allowed to set fishing lines by day or night, provided they use a bird-scaring line.

Table 2.1. Summary of amendments to seabird bycatch mitigation measures in South African permit conditions for foreign-flagged vessels targeting tunas and South African vessels targeting swordfish within the South African EEZ.

Mitigation measure	2006	2007	2008	2009	2010	2011	2012	2013
Foreign-flagged vessels	-	-	-	-	-	-	-	-
Night setting only	NA	Yes	Yes	Yes	Yes	Yes	Yes[A]	Yes[B]
Bird-scaring line	NA	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Line weighting (achieving 0.3 $m.s^{-1}$)	NA	Yes	Yes	Yes	No	No	No	No
Line weighting (60 g < 2m from hook)	NA	No	No	No	Yes	Yes	Yes	Yes
Thawed bait before setting	NA	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Reduced lighting	NA	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Offal management	NA	Yes	Yes	Yes	No	No	No	No
25 bird bycatch limit per year	NA	No	Yes	Yes	Yes	Yes	Yes	Yes
South African vessels	-	-	-	-	-	-	-	-
Night setting only	No	No	No	No	No	No	No	No
Bird-scaring line	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Line weighting (achieving 0.3 $m.s^{-1}$)	Yes	Yes	Yes	Yes	No	No	No	No
Line weighting (60 g < 2m from hook)	No	No	No	No	Yes	Yes	Yes	Yes
Thawed bait before setting	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Reduced lighting	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Offal management	Yes	Yes	Yes	Yes	No	No	No	No
25 bird bycatch limit per year	No	No	Yes	Yes	Yes	Yes	Yes	Yes

[A] Daytime setting is permitted for one vessel at any given time within the South African EEZ, provided that the vessel has obtained prior permission, and ensuring that the vessel uses line weighting and flies two bird-scaring lines.

[B] Daytime setting is permitted for vessels fishing in international waters (beyond the South African EEZ), provided that the vessel has obtained prior permission from the Department of Agriculture, Forestry and Fisheries, and ensuring that the vessel uses line weighting and flies two bird-scaring lines.

In 2006 the Albatross Task Force (ATF) was created to reduce seabird bycatch in fisheries. The first team was hosted by BirdLife South Africa, which commenced work on the deep-sea hake trawl fishery, pelagic longline fishery and hake longline fleet. The ATF was involved in implementing new permit conditions after the termination of foreign bilateral agreements in 2005 meant that no Asian-flagged vessels were issued with South African fishing permits in 2006. Permits were re-issued in 2007 to Japanese and Korean vessels operating under joint venture agreements and required scientific observers to collect data related to fishing operations, including catch and bycatch (DEAT 2008). In 2008 permit regulations were changed to include vessel-specific seabird bycatch limits. Vessels catching 25 birds in a calendar year were required to return to port for inspection of mitigation measures (e.g. adequately designed bird-scaring lines), and to stop fishing for the remainder of the year, unless they were able to demonstrate that they complied with seabird bycatch permit regulations (DEAT 2008). The permits called for fishing to cease if a further 25 birds

were caught, and only resume with a researcher onboard to assess why bycatch rates were so high, but this regulation was not enforced.

Here I estimate seabird bycatch associated with the pelagic longline fishery operating off southern Africa for the period 2006–2013, updating the previous assessment for the period 1998–2005 (Petersen et al. 2009a). I investigate the effect of environmental and vessel-specific variables on seabird bycatch, extrapolate the number of birds killed by unobserved fishing sets, and provide recommendations for further improvements in seabird bycatch regulations. This information could prove important in the adoption of seabird bycatch mitigation requirements by relevant RFMOs. The combination of mandatory night setting for the foreign-flagged (Asian) fleet, and the prevalence of white-chinned petrels in South African waters, a species known to forage efficiently at night (Barnes et al. 1997; Jiménez et al. 2009), leads me to hypothesise that (1) seabird bycatch rates differed between the Asian and South African fleets, and (2) that lunar luminance, which facilitates nocturnal foraging, affects seabird bycatch within this fishery.

2.2 Methods

Data were collected by independent fishery observers onboard South African- and Asian-flagged pelagic longline vessels from 2006–2013. From here on the 'South African pelagic longline fishery' refers to the combined Asian- and South African-flagged fleets. These data are primarily from within the South African EEZ, but also include data from vessels which obtained South African fishing permits but also fished in adjacent international waters. Permit regulations stipulate that any seabirds killed are frozen and returned to port, although compliance with this requirement on vessels without observers is low (c. 60%; Petersen et al. 2009a). Post-mortem examinations were conducted on all returned carcasses to determine species, age and sex (by examination of the gonads). Two closely related species pairs – giant petrels (*M. halli/giganteus*) and royal albatrosses (*Diomedea sandfordi/epomophora*) – could not always be identified reliably to species level and are thus lumped. Molecular analysis confirmed the species identity of 253 shy-type albatrosses (shy *T. cauta* and white-capped *T. steadi*). The remaining shy-type albatrosses killed in the fishery could not be identified to species level and so were also lumped.

Those birds for which only heads were returned to port could not be sexed with confidence and so were excluded from analyses of sex ratios. Chi-squared tests with Yates' correction for continuity were used to test for any deviation from 50:50 sex ratios. For those species for which sex ratios differed from parity, chi-squared tests were performed to test for sex bias in two different age

classes; immatures (including juveniles and subadults) and adults. Age classes were determined for albatrosses using a combination of bill colour, plumage variation and feather wear/moult, and for white-chinned petrels by the amount of scarring on bill plates, moult pattern and gonad development (Hockey et al. 2005). Breeding status of adult birds was determined by presence of a brood patch, size of testes and ovarian follicles.

To assess the likelihood of variables affecting a bycatch event, fishing sets were scored as '1' (where birds were killed) or '0' (no birds killed). Only birds recorded as dead were used in these models as the few birds which were released alive were most likely hooked during hauling (Gilman et al. 2014). Generalised linear models (GLMs) with a binomial distribution and a probit link function were used to test the effect of a number of possible explanatory variables on the likelihood of seabird bycatch occurring (Li 2016). To assess whether variables affected the numbers of birds killed per set, GLMs with a Poisson distribution and a logarithmic link function were used on sets where seabird bycatch was ≥ 1 (Petersen et al. 2009b).

The possible explanatory variables investigated were: year, season, time of setting, branchline length, wind speed, fishing area, lunar luminance and use of a bird-scaring line. Season was classified as summer (December–February), autumn (March–May), winter (June–August) or spring (September–November), although little fishing occurred in summer. Time of setting was classified as day (lines set after nautical sunrise and finished before nautical sunset), night (lines set after nautical sunset and finished before nautical sunrise), or twilight (sets that straddled nautical sunrise/sunset). Wind speed was recorded by fisheries observers at the commencement of setting, and scored using the Beaufort scale (0–8). Fishing areas were divided into international waters (beyond the South African EEZ) and five regions within the South African EEZ (see Figure 2.1). Lunar luminance was calculated as the proportion of overlap of moon presence during setting operations, multiplied by moon phase (0–1) to give a score between 0 and 1, with zero meaning no moon influence during setting and 1 the entire setting process occurring under a full moon. However I was unable to control for cloud cover. Bird-scaring lines were only recorded as 'used' if they were deployed before commencement of line setting and used until completion of line setting. Vessels used either one or two bird-scaring lines simultaneously; however, for simplicity, neither the number of bird-scaring lines deployed nor their design were incorporated into analyses. To control for fishing effort the number of hooks deployed was used as an offset term in the models. Sets with incomplete explanatory (8.1% of observed sets), were excluded from the GLM analyses. The most appropriate model was selected using Akaike's information criterion (AIC) as well as the proportion of variance explained by the models.

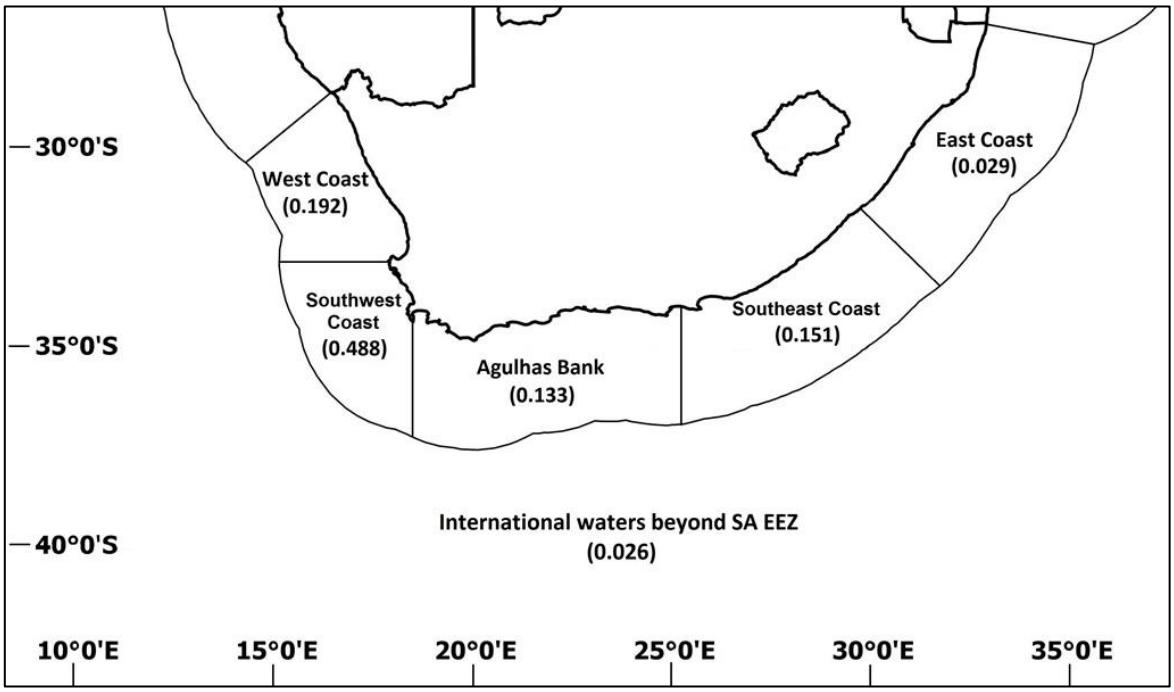


Fig. 2.1. The South African EEZ showing the areas used in models of seabird bycatch (GLMs), with associated catch rates (birds per 1000 hooks). The seabird bycatch rate indicated for international waters beyond the South African EEZ only includes vessels which obtained South African fishing permits.

Due to large differences in gear configuration and operations between the Asian and South African fleets, separate GLMs were used to test the influence of variables affecting seabird bycatch for each fleet. GLMs were also used to test the effect of variables on the likelihood and amount of bycatch associated with Asian-flagged vessels for the four most commonly recorded species/species groups: white-chinned petrels, shy-type albatrosses, black-browed albatrosses and yellow-nosed albatrosses. Too few birds were reported caught by South African-flagged vessels to run similar analyses. The two yellow-nosed albatrosses were lumped because they were not differentiated by fisheries observers, and only 51% of those caught (n = 186) were available for necropsy (Table 2.2). I used linear extrapolations from observed trips to estimate overall mortality on unobserved trips by South African-flagged vessels for two periods (pre- and post-2008) by area (5 × 5° square) and season. Values given are means ± standard deviation, unless otherwise stated.

Table 2.1. Species composition of seabird bycatch from both the South African and Asian pelagic longline fleets off South Africa, 2006–2013 as reported by fisheries observers and confirmed by autopsies.

Common Name	Scientific Name	IUCN Threat Status	Reported		Confirmed	
			n	%	*n*	%
Shy/white-capped albatross	*Thalassarche cauta/steadi*	Near-Threatened	508	17.8	482	20.5
Black-browed albatross	*Thalassarche melanophris*	Near-Threatened	130	4.6	159	6.8
Indian yellow-nosed albatross	*Thalassarche carteri*	Endangered	-	-	77	3.3
Atlantic yellow-nosed albatross	*Thalassarche chlororhynchos*	Endangered	-	-	18	0.8
Yellow-nosed albatross sp.	*Thalassarche carteri/chlororhynchos*	-	186	6.5	-	-
Northern/southern royal albatross	*Diomedea sandfordi/epomophora*	Endang/Vuln	2	< 0.1	3	0.1
Wandering albatross	*Diomedea exulans*	Vulnerable	10	0.4	5	0.2
Unidentified albatrosses	-	-	65	2.3	-	-
Northern/southern giant petrel	*Macronectes halli/giganteus*	Least Concern	16	0.6	7	0.3
White-chinned petrel	*Procellaria aequinoctialis*	Vulnerable	1768	62.0	1541	65.7
Grey petrel*	*Procellaria cinerea*	Near-Threatened	1	< 0.1	1	< 0.1
Cape petrel*	*Daption capense*	Least Concern	2	< 0.1	1	< 0.1
Unidentified petrels	-	-	84	2.9	-	-
Great shearwater	*Ardenna gravis*	Least Concern	2	< 0.1	2	0.1
Brown skua	*Stercorarius antarctica*	Least Concern	3	0.1	2	0.1
Cape gannet	*Morus capensis*	Vulnerable	73	2.6	45	1.9
King penguin*	*Aptenodytes patagonicus*	Least Concern	1	< 0.1	1	< 0.1
Total			2851	100	2344	100

* First necropsy records from the South African pelagic longline fishery (although not all caught in South African EEZ)

2.3 Results

2.3.1 Fishery characteristics

During the period 2006–2013, ~35 million hooks (17 448 sets) were set by 66 vessels: 29 Asian-flagged vessels targeting tuna (67% total effort) and 37 South African-flagged vessels targeting swordfish (33% total effort). Observer coverage was 100% on Asian vessels but variable (average of 6.2% p.a.) on South African vessels with 10.9 million hooks unobserved (31% of total effort, and 94% of effort by South African vessels). Fishing effort was greatest in 2011 with a combined total of 6.4 million hooks set by both fleets, whereas only 1.1 million hooks were set by South African-flagged vessels in 2006 (with zero Asian-flagged fishing effort in 2006). Asian-flagged vessels set an average of 3.3 million hooks per year, whereas South African vessels set an average of 1.4 million hooks per year for the same period (Figure 2.2). Asian fishing effort peaked between April and October (Figure 2.2), with fishing conducted throughout the South African EEZ and regularly venturing farther into international waters beyond the South African EEZ (Figure 2.3a). South African vessels fished

throughout the year (Figure 2.2), concentrating fishing effort within the South African EEZ (Figure 2.3b).

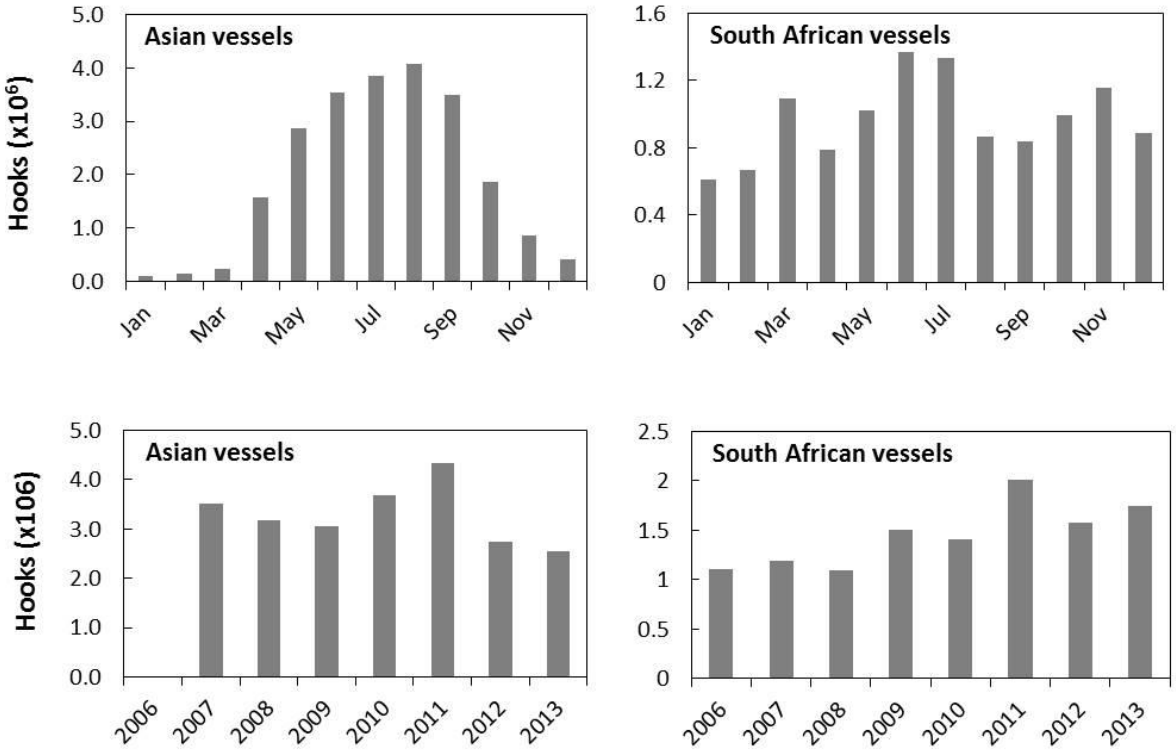
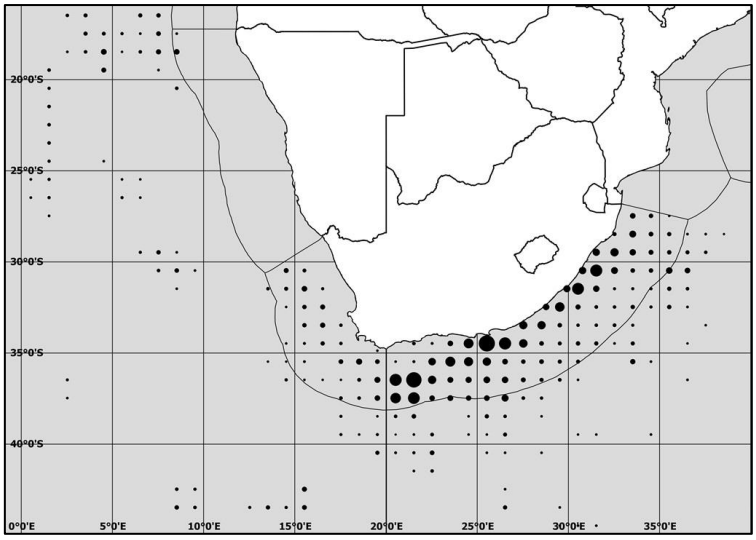


Fig. 2.2. Number of hooks set according to month and year for Asian and South African pelagic longline vessels, 2006–2013, fishing within the South African EEZ and adjacent international waters.

a.



b.

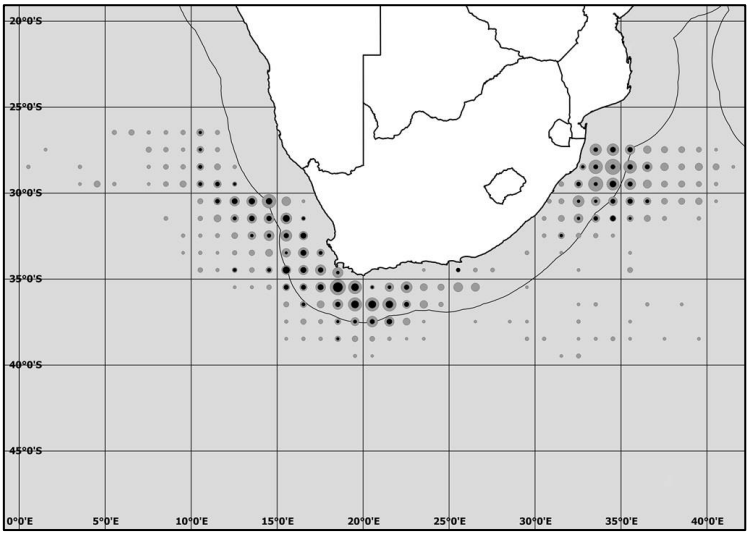


Fig. 2.3. Distribution of fishing effort of a. Asian and b. South African vessels off southern Africa, in relation to national EEZs, 2006–2013. Grey circle size is a proportional measure of number of hooks set per 1 ^{o} square, with black circle size representing a proportional measure of observer coverage. Asian vessels had 100 % observer coverage.

Due to differences in target species (Figure 2.4), Asian vessels use longer branchlines (36 ± 4.7 m; 93% of sets had branchlines > 30 m) than South African vessels (21 ± 9.1 m; 82% of sets had branchlines < 30 m), and unlike South African vessels do not use light-sticks. South African vessels (1 300 ± 240 m) averaged fewer hooks per set than Asian vessels (2 700 ± 380 m) and typically set their lines at night (89%) whereas South African vessels set mostly around sunset (68%; Figure 2.5).

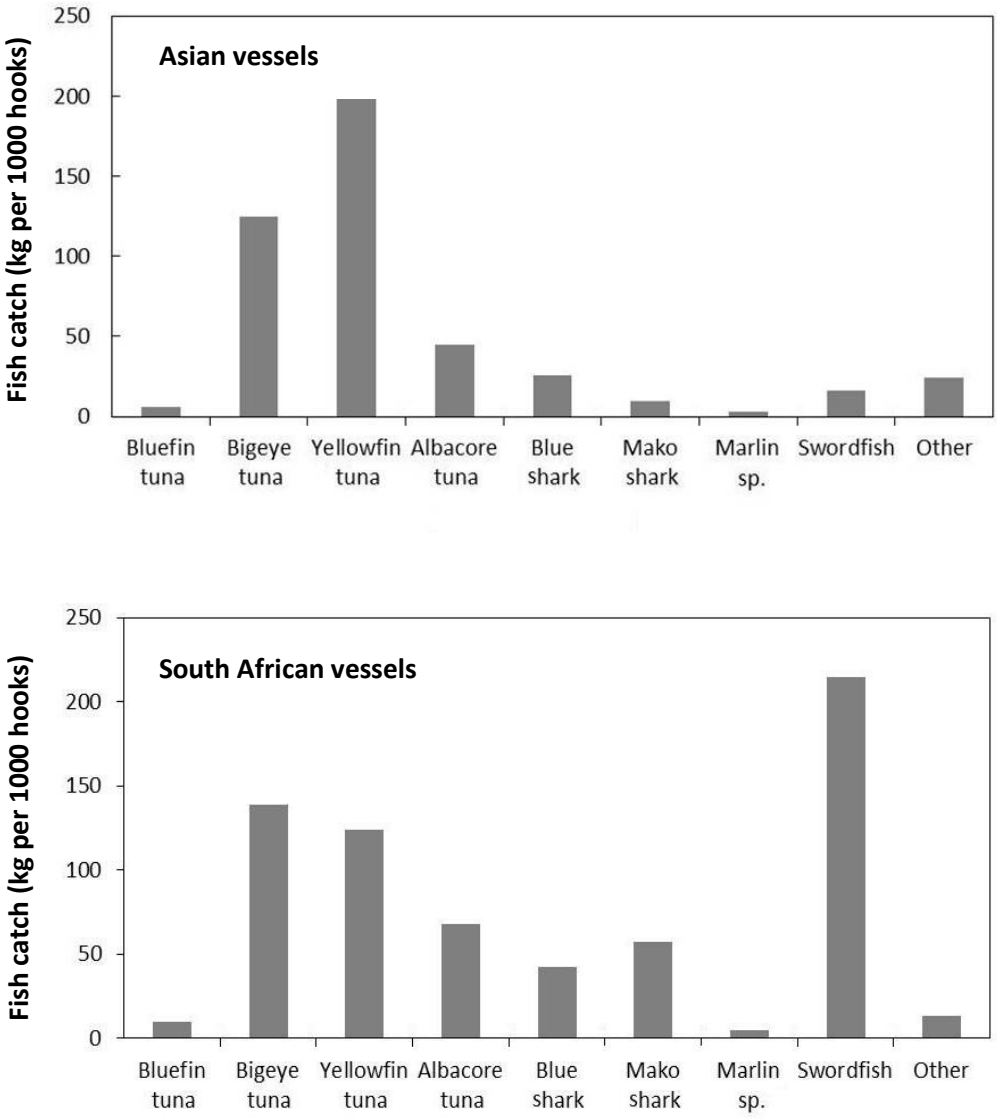


Fig. 2.4. Fish catch for both Asian and South African pelagic longline vessels, 2006–2013.

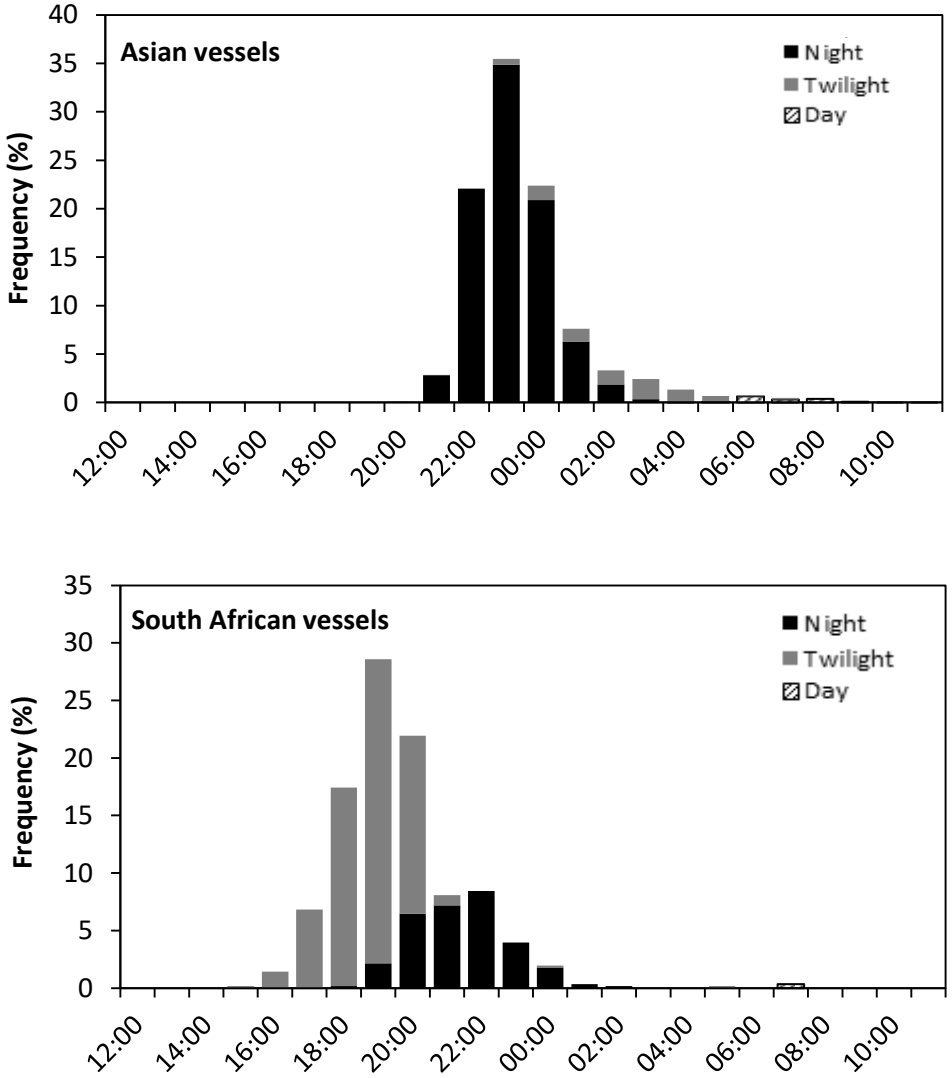


Fig. 2.5. Time of commencement of setting for Asian and South African vessels, 2006–2013. See methods for definition of night, twilight and day sets.

2.3.2 Seabird bycatch

From 2006–2013, 2 851 seabird mortalities were recorded as bycatch in the (observed) pelagic longline fishery off South Africa, of which 2 345 (83%) were returned to port for necropsy. A further 307 birds (11% of total reported bycatch mortality) were caught alive and released. Bycatch included 14 species, of which seven are listed as threatened by the IUCN (Table 2.2). Post-mortems confirmed that white-chinned petrels were most frequently caught, comprising 66% of total bycatch. Collectively the five albatrosses commonly killed represented 32% of seabird bycatch: shy-type 21%, black-browed 7%, Indian yellow-nosed 3% and Atlantic yellow-nosed 1% (Table 2.2). The remaining 2% comprised an additional nine species/species groups, dominated by Cape gannets (Table 2.2).

From 2006 to 2009, the bycatch rate was 0.148 birds per 1 000 hooks; from 2010 to 2013, the bycatch rate was 0.07 birds per 1 000 hooks.

The two most commonly killed species returned to port showed significant departures from the expected 50:50 sex ratio. White-chinned petrels exhibited a slight male-bias (53%; $\chi^2 = 5.6$; df = 1; $p = 0.02$; Table 2.3) whereas mortality of shy-type albatrosses was female-biased (57%; $\chi^2 = 7.3$; df = 1; $p < 0.01$). This was driven by sex-biased mortality of immature birds (male biased white-chinned petrels; 58%; $\chi^2 = 17.8$; df = 1; $p < 0.01$, female biased shy-type albatrosses; 57%; $\chi^2 = 5.8$; df = 1; $p = 0.02$), because no sex bias was found for adults of either species.

Table 2.3. Age composition and sex ratios (%F = % Female) of the most commonly recorded seabird bycatch species off South Africa. Age and sex were determined by post-mortem analysis. Breeding adults are expressed as a percentage of adults caught.

Species	**% Imm (% F)**	**% Adult (% F)**	**% Br. adult (% F)**	**Total (% F)**
White-chinned petrel	51 (42)	49 (51)	10 (5)	47
Shy-type albatrosses	84 (57)	16 (53)	3 (50)	57
Black-browed albatross	78(50)	23 (57)	11 (0)	53
Indian yellow-nosed albatross	25 (39)	75 (35)	14 (25)	44
Atlantic yellow-nosed albatross	11 (100)	89 (53)	41 (43)	57
Cape gannet	0 (-)	100 (47)	40 (26)	47
Total	57 (55)	43 (51)	12 (13)	49

Of the autopsied birds for which age could be determined, most were immatures (57%; Table 2.3), but this varied greatly among species. Immatures dominated for both shy-type and black-browed albatrosses. Similar numbers of immatures and adults were recorded for white-chinned petrels, whereas most yellow-nosed albatrosses (both species) and all Cape gannets killed were adults (Table 2.3).

Of the 739 adult white-chinned petrels autopsied, 73 (10%) had either brood patches or enlarged gonads, which would suggest breeding or a recently abandoned breeding attempt. In total, 41% (n = 17) of the adult Atlantic yellow-nosed albatrosses and 40% (n = 45) of the Cape gannets caught showed signs of breeding activity. All Cape gannets were killed within c. 250 km of the coast, with 72% killed off the west and southwest coasts and the remainder on the Agulhas Bank and off the southeast coast (see Figure 2.1 for regions).

Between 2006 and 2013 observed seabird bycatch from Asian- and South African-flagged vessels totalled 2 743 and 108 birds, respectively (Table 2.4). Asian-flagged vessels caught 343 birds per year between 2006 and 2013; seabird bycatch was high between 2006 and 2009 (458 birds per year), but was reduced to 228 birds per year between 2010 and 2013. South African vessels averaged 22 birds per year between 2006 and 2009, with no observer coverage between 2010 and 2013. Extrapolation from the observed bycatch to the 2 256 unobserved sets from South African-flagged vessels suggests that an additional 94 birds (SE = 17.3) were caught per year by this fleet (Table 2.4). Combined total seabird bycatch for both fleets (recorded and extrapolated) averaged 451 birds per year between 2006 and 2013. However, bycatch was reduced to 361 birds per year between 2010 and 2013 (Table 2.4) from 563 birds per year between 2006 and 2009.

Table 2.4. Summary of yearly bycatch of the four most commonly recorded bycatch species and all species recorded as bycatch. The figures in bold represent the entire bycatch for the fishery, with Asian bycatch/South African bycatch/extrapolated bycatch included.

Year	Shy-type albatrosses	Black-browed albatross	Yellow-nosed albatrosses	White-chinned petrel	All species
2006	**45** (0/16/29)	**2** (0/0/2)	**0** (0/0/0)	**11** (0/3/8)	**76** (-/29/47)
2007	**279** (236/7/36)	**23** (15/3/5)	**87** (87/0/0)	**657** (638/5/14)	**1253** (1171/15/67)
2008	**132** (79/6/47)	**18** (12/1/5)	**9** (8/0/1)	**107** (103/2/2)	**236** (167/9/60)
2009	**90** (39/17/34)	**46** (33/3/10)	**15** (10/1/4)	**365** (360/1/4)	**597** (495/35/67)
2010	**61** (33/0/28)	**22** (7/5/10)	**15** (12/1/2)	**147** (142/1/4)	**293** (207/20/66)
2011	**73** (21/-/52)	**25** (8/-/17)	**66** (64/-/2)	**274** (266/-/8)	**602** (416/-/186)
2012	**32** (11/-/21)	**7** (1/-/6)	**1** (0/-/1)	**118** (115/-/3)	**269** (137/-/132)
2013	**62** (0/-/62)	**8** (0/-/8)	**5** (3/-/2)	**135** (132/-/3)	**278** (150/-/128)
Total	**774** (419/46/309)	**151** (76/12/63)	**198** (184/2/12)	**1814** (1756/12/46)	**3604** (2743/108/753)
Average (2006–09)	**137** (89/12/37)	**22** (15/2/6)	**28** (26/1/1)	**285** (275/3/7)	**563** (458/22/60)
Average (2010–13)	**57** (16/-/41)	**16** (4/-/10)	**22** (16/-/2)	**169** (164/-/5)	**361** (228/-/128)
Average (2006–13)	**97** (52/9/39)	**19** (9/2/8)	**25** (23/1/2)	**227** (220/2/6)	**451** (343/22/94)

2.3.3 Effects of vessel flag, vessel identification and branchline length

In models including vessel flag but not vessel identification or branchline length, flag was an important variable affecting the numbers of seabirds killed ($p < 0.001$); however, it did not affect the likelihood of a bycatch event occurring ($p = 0.434$). Asian-flagged vessels caught birds at a lower rate (0.129 birds per 1 000 hooks) than South African vessels (0.209 birds per 1 000 hooks), although the much greater Asian fishing effort and observer coverage meant that these vessels accounted for 96% of recorded seabird bycatch during the study period. Seabird bycatch occurred throughout the subregion for Asian-flagged vessels, but was restricted to the west coast, southwest coast and Agulhas Bank for South African-flagged vessels (Figure 2.6).

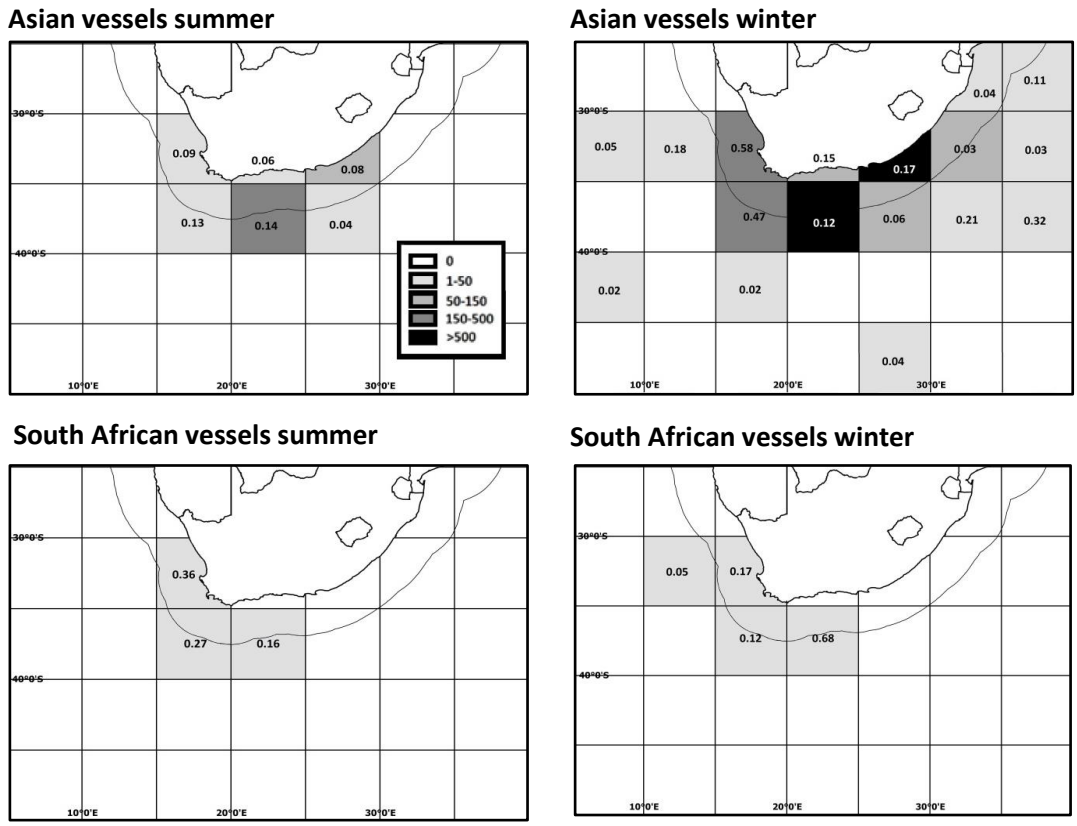


Fig. 2.6. Number (denoted by shades of grey) and rate (birds per 1000 hooks) of birds caught by Asian and South African vessels from May–October and November–April, divided into 5x5 ^{o} squares.

In models where vessel identification (both fleets) was included but vessel flag and branchline length excluded, vessel identification was an important factor affecting both the likelihood of bycatch ($p < 0.001$) and the numbers of seabirds killed ($p = 0.002$). Five Asian vessels (setting c. 40% of total hooks) accounted for > 50% of total seabird mortality. Of these vessels, one vessel (setting c. 7% of total hooks) caught > 330 birds over six years (11% of total seabird bycatch) at a rate of 0.200 birds per 1 000 hooks, although that rate varied substantially across years.

Branchline length affected both the likelihood ($p = 0.004$) and numbers of seabird bycatch ($p < 0.001$; Table 2.5) for the Asian fleet, with bycatch increasing with branchline length. However, this variable did not vary significantly for the South African fleet, where branchlines are shorter and for which there were fewer data.

Table 2.5. The effect of different variables on seabird bycatch for Asian vessels, based on the best fit GLM. Significant variables are highlighted in bold.

Variable	Estimate	SE	z-value	*p*
Asian: bycatch likelihood	-	-	-	-
Intercept	**220.6**	**26.81**	**8.227**	**< 0.001**
Year	**-0.115**	**0.013**	**-8.578**	**< 0.001**
Branchline length	**< 0.001**	**< 0.001**	**2.901**	**0.004**
Time of setting (night)	-0.269	0.261	-1.033	0.301
Time of setting (sunrise)	0.259	0.257	1.011	0.312
Time of setting (sunset)	-5.059	0.028	-0.018	0.985
Lunar luminance	**1.043**	**0.060**	**17.515**	**< 0.001**
Season (spring)	-0.046	0.091	-0.504	0.614
Season (summer)	**0.907**	**0.326**	**2.785**	**0.005**
Season (winter)	-0.030	0.077	-0.386	0.699
Area (southwest coast)	-4.761	192.6	-0.025	0.980
Area (east coast)	**-0.902**	**0.240**	**-3.765**	**< 0.001**
Area (international waters)	-0.300	0.387	-0.776	0.438
Area (southeast coast)	**-0.296**	**0.102**	**-2.895**	**0.004**
Area (west coast)	-3.763	349.4	-0.011	0.991
Bird-scaring line	**-0.263**	**0.103**	**-2.548**	**0.011**
Interaction (spring x southwest coast)	5.069	192.6	0.026	0.979
Interaction (summer x southwest coast)	4.252	192.6	0.022	0.982
Interaction (winter x southwest coast)	5.225	192.6	0.027	0.978
Interaction (spring x east coast)	0.215	0.286	0.752	0.452
Interaction (summer x east coast)	-4.697	56.35	-0.083	0.934
Interaction (winter x east coast)	0.478	0.263	1.816	0.069
Interaction (spring x international waters)	-0.436	0.418	-1.043	0.297
Interaction (summer x international waters)	-5.525	64.65	-0.085	0.932
Interaction (winter x international waters)	**-0.879**	**0.438**	**-2.008**	**0.045**
Interaction (spring x southeast coast)	**0.335**	**0.140**	**2.390**	**0.017**
Interaction (summer x southeast coast)	-5.235	114.0	-0.046	0.963
Interaction (winter x southeast coast)	**0.386**	**0.119**	**3.243**	**0.001**
Interaction (spring x west coast)	3.936	349.4	0.011	0.991
Interaction (summer x west coast)	2.023	349.4	0.006	0.995
Interaction (winter x west coast)	4.607	349.4	0.013	0.989
Asian: bycatch numbers	-	-	-	-
Intercept	36.16	28.38	1.275	0.202
Year	-0.022	0.014	-1.558	0.119
Branchline length	**< 0.001**	**< 0.001**	**6.423**	**< 0.001**
Time of setting (night)	-0.099	0.321	-0.308	0.758
Time of setting (sunrise)	0.333	0.325	1.025	0.306
Lunar luminance	**0.354**	**0.061**	**5.770**	**< 0.001**
Season (spring)	**0.275**	**0.075**	**3.684**	**< 0.001**
Season (summer)	**-0.897**	**0.233**	**-3.851**	**< 0.001**
Season (winter)	**0.233**	**0.065**	**3.592**	**< 0.001**
Area (southwest coast)	**0.480**	**0.105**	**4.563**	**< 0.001**
Area (east coast)	**-0.262**	**0.121**	**-2.170**	**0.030**
Area (international waters)	**-1.109**	**0.176**	**-6.303**	**< 0.001**
Area (southeast coast)	**0.141**	**0.045**	**3.130**	**0.002**
Area (west coast)	-0.306	0.176	-1.741	0.082
Bird-scaring line	**-0.550**	**0.080**	**-6.849**	**< 0.001**

2.3.4 Effect of time of setting, lunar luminance and sea state

Time of setting did not significantly affect seabird bycatch rates for either fleet. Despite this, overall the highest bycatch rates occurred when setting took place around sunrise (0.232 birds per 1 000 hooks). Surprisingly the lowest bycatch rates occurred during daylight sets (0.022 birds per 1 000 hooks). However, 86% of day sets were in international waters, where seabird abundance is lower than along the continental shelf, which may have affected seabird bycatch rates.

Lunar luminance influenced bycatch rates significantly for both Asian- and South African-flagged vessels. For the Asian fleet, increasing lunar luminance increased both the likelihood and the numbers of birds killed ($p < 0.001$; Table 2.5) while for the South African fleet, only the numbers of birds killed increased with lunar luminance ($p = 0.032$; Table 2.6). Overall bycatch rates peaked (0.378 birds per 1 000 hooks) when lunar luminance was greatest (> 0.9), and were more than eight times higher than during periods of lowest (< 0.1) lunar luminance (0.047 birds per 1 000 hooks; Figure 2.7).

Table 2.6. The effect of different variables on seabird bycatch for South African vessels, based on the best fit GLM. Significant variables are highlighted in bold.

Variable	Estimate	SE	z-value	*p*
South African: bycatch likelihood	-	-	-	-
Intercept	**-8.404**	**0.311**	**-27.012**	**< 0.001**
Season (spring)	**-0.988**	**0.460**	**-2.148**	**0.032**
Season (summer)	-0.334	0.260	-1.284	0.199
Season (winter)	0.205	0.295	0.697	0.486
Area (southwest coast)	**-0.721**	**0.308**	**-2.341**	**0.019**
Area (east coast)	-5.177	271.404	-0.019	0.985
Area (international waters)	-5.171	302.928	-0.017	0.986
Area (southeast coast)	-4.126	1459.236	-0.003	0.998
Area (west coast)	-0.278	0.331	-0.840	0.401
Bird-scaring line	0.415	0.2920	1.421	0.155
South African: bycatch numbers	-	-	-	-
Intercept	**-6.924**	**0.332**	**-20.872**	**< 0.001**
Lunar luminance	**0.889**	**0.414**	**2.146**	**0.032**

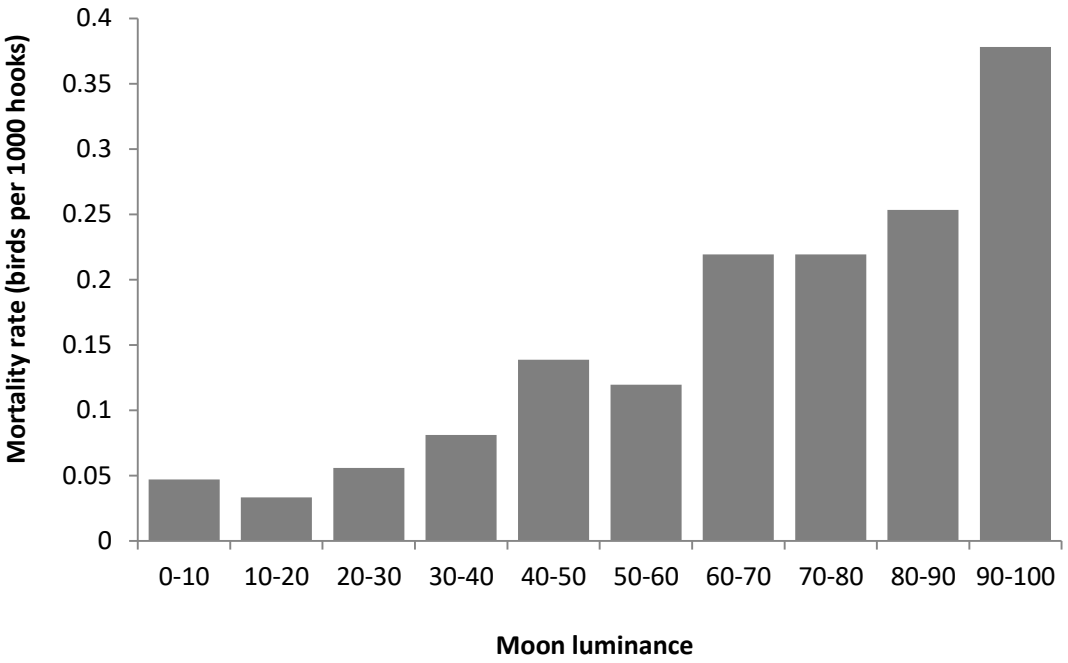


Fig. 2.7. The effect of moon influence on bird mortality on longlines set at night off South Africa, 2006–2013. Moon influence was calculated as the proportion of overlap of moon presence during setting operations, multiplied by moon phase (0–1) to give a score between 0 and 1; zero meaning no moon influence during setting and 1 the entire setting process occurring under a full moon.

Wind strength and resultant sea state did not have any influence on either the likelihood or numbers of total seabird bycatch in the Asian and South African fleets. It did, however, influence bycatch of yellow-nosed albatrosses within the Asian fleet, with both the likelihood ($p = 0.022$) and numbers ($p < 0.001$) of bycatch decreasing with increasing wind strength.

2.3.5 Effect of year, season and area

Between 2006 and 2013, birds were caught at an average rate of 0.132 birds per 1 000 hooks (foreign vessels = 0.129 per 1 000 hooks, local vessels = 0.209 per 1 000 hooks). However, this rate varied substantially throughout the study period. Year was an important variable influencing the likelihood of seabird bycatch in the Asian fleet ($p < 0.001$; Table 2.5). Bycatch was greatest in 2007 (0.333 and 0.350 birds per 1 000 hooks for Asian and South African vessels, respectively) but then decreased following the changes to permit regulations in 2008 that placed vessel-specific limits on seabird catches. Bycatch rates during 2008–2013 averaged 0.080 and 0.141 birds per 1 000 hooks for Asian and South African vessels, respectively (Figure 2.8).

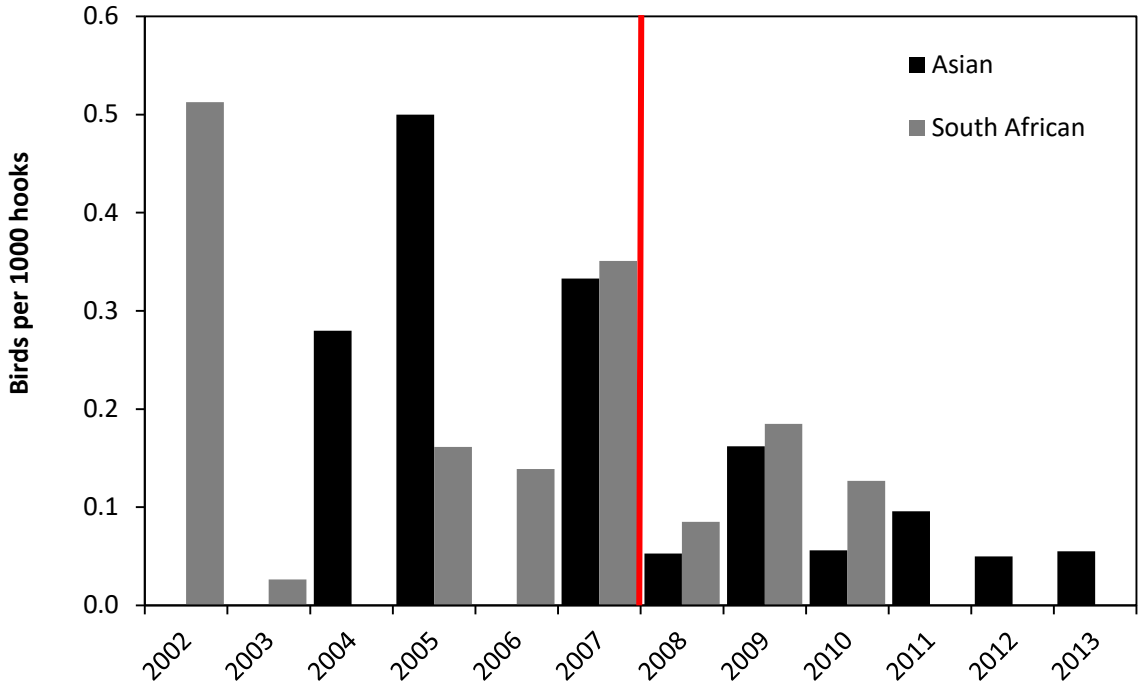


Fig. 2.8. Seabird mortality for the study period (2006–2013) as well as the years preceding the study (2002–2005) for observed Asian and South African tuna longline vessels, expressed as birds per 1000 hooks. No observer data exists for South African vessels for 2004 and 2011–2013, while no observer data exists for Asian vessels in 2002, with zero fishing effort in 2003 and 2006. Vessel-specific limits on seabird bycatch began in 2008 (denoted by the red line).

Season affected both the likelihood and numbers of birds killed for the Asian fleet but only the likelihood of capture for South African vessels (Tables 2.4 and 2.5). Overall, birds were caught at higher rates during winter (0.128 birds per 1 000 hooks) and spring (0.110 birds per 1 000 hooks) than during autumn (0.070 birds per 1 000 hooks) and summer (0.053 birds per 1 000 hooks). Similar seasonal trends occurred for the bycatch of the four most commonly caught species/species groups, for both fleets combined (Figure 2.9).

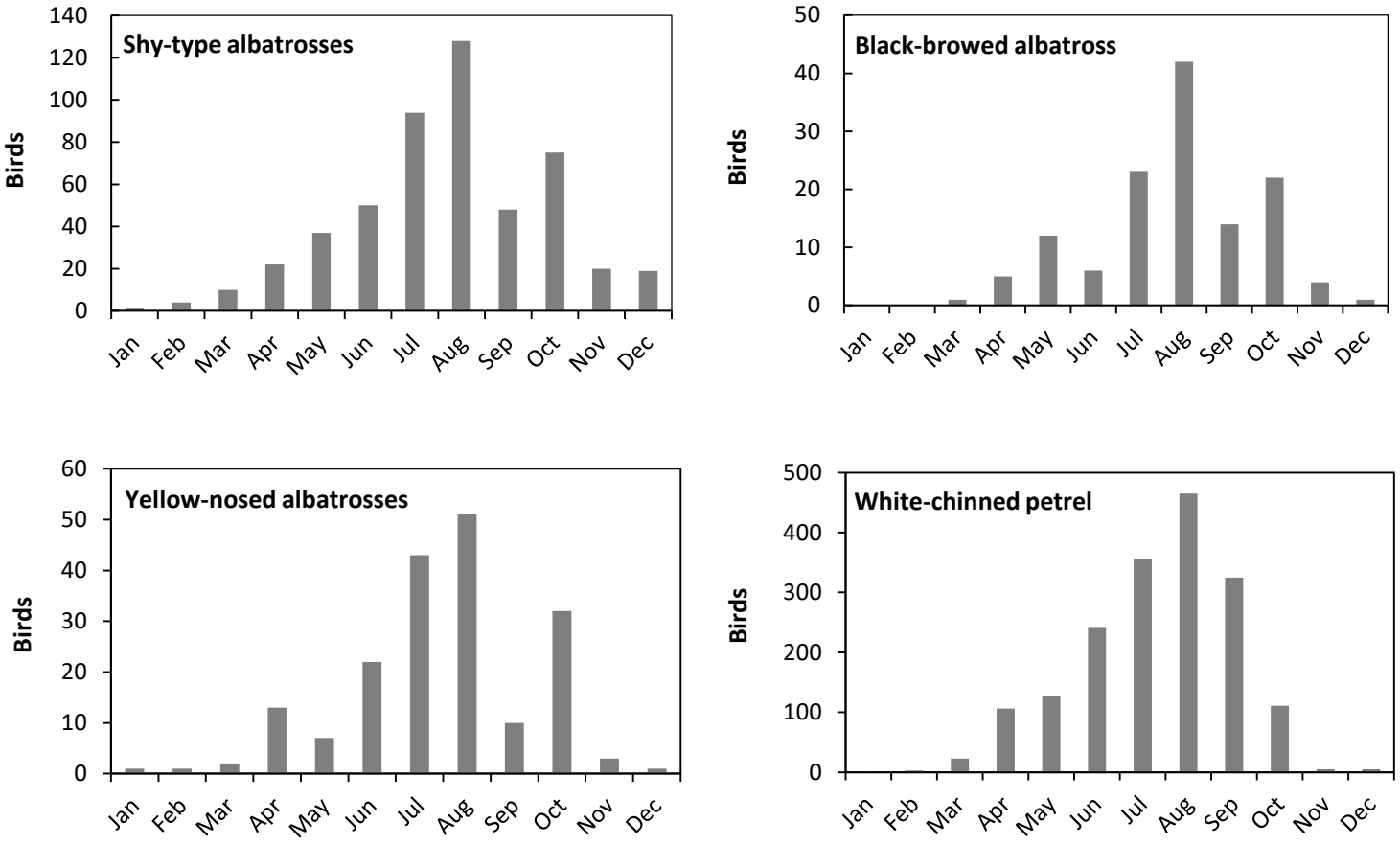


Fig. 2.9. Mortality of the four most commonly caught species/species groups; shy-type, black-browed and yellow-nosed albatrosses and white-chinned petrels according to month, for both the South African and Asian fleets combined.

Area influenced both the likelihood and numbers of seabirds killed by Asian vessels and had a strong influence on the likelihood of seabird bycatch occurring for South African vessels. For Asian vessels, significant differences were found in the number of birds killed between the Agulhas Bank and most other areas, with the highest bycatch rate recorded along the southwest coast ($p < 0.001$; Table 2.5). For South African vessels the likelihood of a bycatch event was reduced when fishing off the southwest coast ($p = 0.019$; Table 2.6) compared to the Agulhas Bank. Overall, birds were caught at the greatest rates off the southwest coast (0.488 birds per 1 000 hooks) and west coast (0.192 birds per 1 000 hooks), while the lowest rates occurred off the east coast and in international waters (both 0.030 birds per 1 000 hooks; Figure 2.1), with similar results for the four most commonly recorded species/species groups (Figure 2.10).

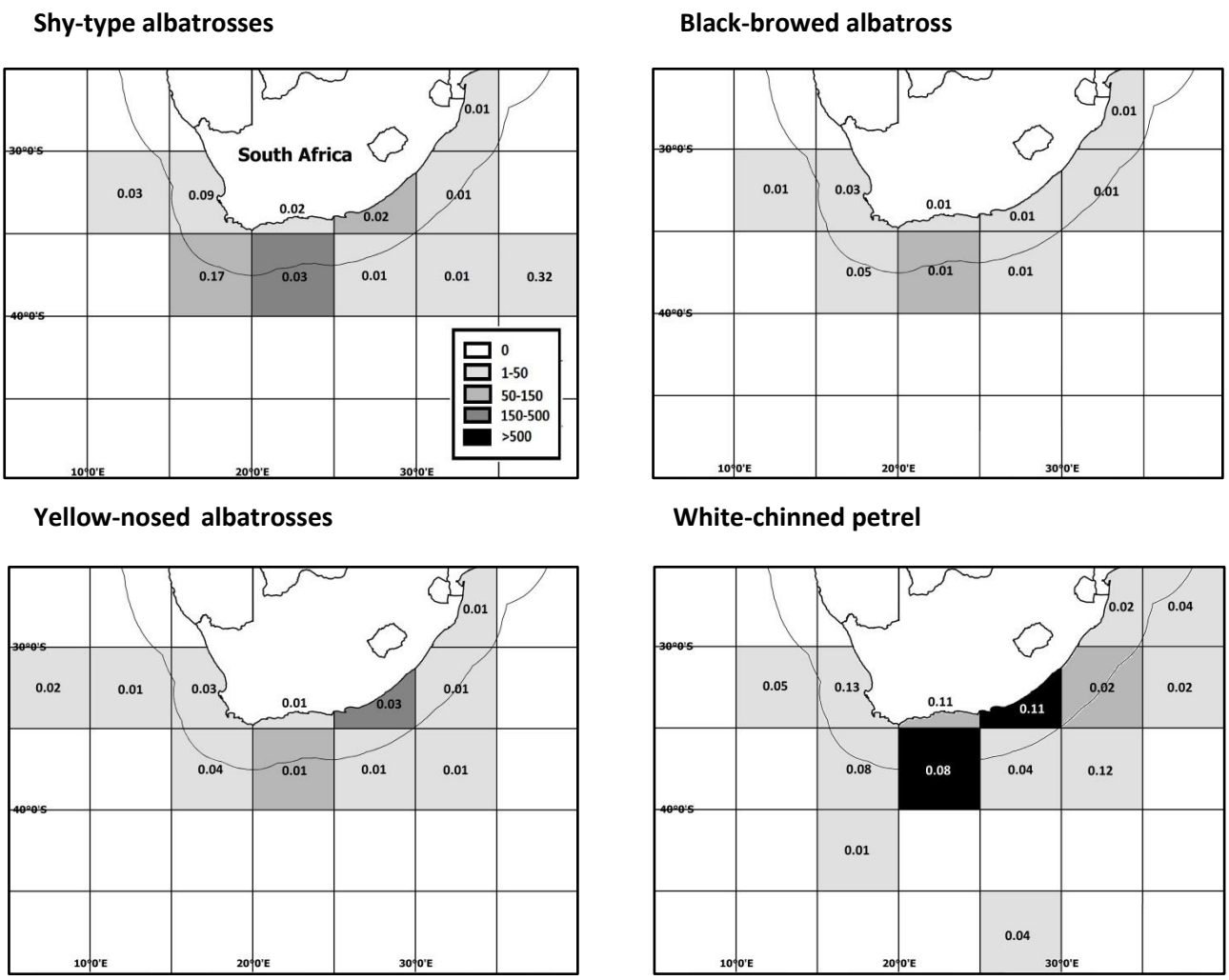


Fig. 2.10. Numbers (denoted by shades of grey) and rates (birds per 1000 hooks) of the four most commonly caught species/species groups divided into 5 x 5^{o} squares, for both the South African and Asian fleet.

The interaction between season and area affected the likelihood of seabird bycatch on Asian vessels only. The likelihood (p = 0.045) of seabirds being killed was reduced when fishing in winter in international waters (Table 2.5), but increased when fishing off the southeast coast in spring (p = 0.017) and winter (p = 0.001).

2.3.6 Effect of bird-scaring lines

The use of bird-scaring lines reduced the likelihood (p = 0.011) and numbers (p < 0.001) of seabirds killed for the Asian fleet (Table 2.5). They did not affect either the likelihood or numbers of seabirds

killed on South African vessels, probably as they were deployed selectively in areas with larger numbers of birds. Correctly deployed bird-scaring lines reduced bycatch threefold on Asian vessels (from 0.333 to 0.110 per 1 000 hooks), however birds were caught at a higher rate on sets which correctly deployed bird-scaring lines on South African vessels (0.010 versus 0.110 birds per 1 000 hooks). Compliance with the requirement to use bird-scaring lines was far greater for Asian vessels than observed South African vessels, and improved with time for both fleets (Table 2.7).

Table 2.7. Summary of bird-scaring line deployment for Asian and observed South African vessels for which bird-scaring line data was available for 2006–2013.

Year	Number of sets observed			Percent with bird-scaring lines		
	Asian	South African	Total	Asian	South African	Total
2006	0	139	139	-	38.8	38.8
2007	1329	37	1366	81.8	48.6	80.9
2008	1148	80	1228	96.4	78.8	95.3
2009	1128	138	1266	99.9	84.0	98.2
2010	1319	121	1440	99.8	81.0	98.3
2011	1557	6	1653	99.9	100	99.9
2012	1014	0	1014	99.7	-	99.7
2013	1003	0	1003	100	-	100
Total	8498	521	9019	96.6	68.1	94.9

2.4 Discussion

2.4.1 Numbers and rates of seabird bycatch

Seabird bycatch rates were generally similar to or lower than those reported by other seabird studies from pelagic longline fisheries in the Southern Hemisphere. South American fisheries recorded higher seabird bycatch rates (range 0.095–5.03 birds per 1 000 hooks; see review by Bugoni et al. 2008), with a slight decrease in bycatch rates in recent years attributable to better sampling coverage rather than improved conservation actions. Rates for the eastern tuna and billfish fishery off Australia (2001–2006; Trebilco et al. 2010) were similar to rates from my study, with bycatch rates lower in later years (2004–2006). Waugh et al. (2008) estimated c. 500 birds were killed per year between 1998 and 2004 (no rate per 1 000 hooks supplied) in the New Zealand pelagic longline fishery. This is a large reduction from 1988–1992 figures (attributable to improved mitigation measures) where an estimated 3 600 birds were killed in one year (Murray et al. 1993).

Seabird bycatch rates in the South African pelagic longline fishery during 2006–2013 (2 851 observed mortalities; 0.132 birds per 1 000 hooks) were 3–12 times lower than bycatch estimates for this fishery prior to 2006 (Figure 2.8; Ryan et al. 2002; Petersen et al. 2009a). However, the most important change occurred in 2008, when vessel-specific bycatch limits were put in place for both South African and Asian vessels. This resulted in a significant drop in bycatch rates for both fleets. From 2010–2013 bycatch rates were reduced to 0.07 birds per 1 000 hooks (range 0.05–0.10 birds per 1 000 hooks). Clearly the presence of observers and the imposition of a meaningful cost for catching large numbers of seabirds changed fishers' behaviour and reduced seabird bycatch. However, low observer coverage, dislodgement of hooked birds from lines (Gilman et al. 2005) and release of live-captured but badly injured birds are all likely to result in mortality higher than recorded in my study.

Fishing effort for the current study period (2006–2013) was slightly lower than the previous study period (1998–2005; Petersen et al. 2009a) for South African-flagged vessels (1.3 million versus 1.4 million hooks per year) but was significantly lower for Asian-flagged vessels (3.3 million versus 5.2 million hooks per year). Fishing effort varied considerably per year in my study; after fishing effort peaked in 2011 (6.3 million hooks), it was considerably reduced in 2012 (4.3 million hooks) and 2013 (4.7 million hooks).

Estimation of seabird bycatch from the unobserved sets deployed by South African vessels, using area and season as two predictive variables, yielded a further 753 birds likely to have been killed between 2006 and 2013. However, this estimate, based on simple extrapolation, should be treated with caution as seabird bycatch on unobserved trips may be higher than on observed trips, due to likely lower compliance levels with bycatch mitigation regulations (Gales 1998).

In other longline fisheries, large numbers of observed birds hooked during setting (as many as 50%, Brothers et al. 2010) become dislodged from the hooks due to fish predation, currents or mechanical actions during the line soak or haul (Gilman et al. 2005). Thus it is possible that a maximum of c. 1 400 birds (50% of total recorded bycatch) were additionally killed by the fishery during line setting, without their carcasses retrieved during line hauling.

An additional 307 live birds were recorded as caught during hauling, but were subsequently dehooked and released. In most cases it is impossible to determine whether these birds survived or not. Gilman et al. (2014) reported that all birds caught during hauling were alive when brought aboard, and I have assumed that live birds caught during my study were caught during hauling operations.

2.4.2 Impact on species

White-chinned petrels are the most frequently caught seabird species off South Africa, accounting for almost two thirds of bycatch in the South African pelagic longline fishery, and are also the most commonly caught species throughout the Southern Hemisphere (Ryan et al. 2012). They are proficient divers, capable of reaching depths of up to 16 m (Chapter 5). It has been suggested that they increase the bycatch of larger, shallower diving species, such as albatrosses, because they raise baits to depths accessible to shallow-diving species, only to be displaced by the larger birds (Jiménez et al. 2012).

A bias in bycatch towards the deeper-diving males (Chapter 5) has previously been found in white-chinned petrel longline bycatch studies (Robertson et al. 2006; Petersen et al. 2009a). Male white-chinned petrels may be caught at greater rates than females, as due to their larger sizes they are able to scavenge more successfully than females (Ryan and Boix-Hinzen 1999). Another possible explanation is that there is sexual segregation in foraging zones, but movements studies have not confirmed this (Berrow et al. 2000a; Chapter 4). A sex bias is of particular concern to seabirds as many species have strong pair-bonds and need both sexes to raise chicks successfully (Delord et al. 2005), thus further exacerbating the effect of seabird bycatch.

The only white-chinned petrel populations known to forage regularly in South African waters are those from PEI (Chapter 4), Crozet Island (Weimerskirch et al. 1999) and Kerguelen Island (Péron et al. 2010), where populations number 36 000 (Ryan et al. 2012), 23 000 (Barbraud et al. 2008) and 234 000 (Barbraud et al. 2009) breeding pairs, respectively. Molecular studies confirm that white-chinned petrels caught by longlines off South Africa are from the subspecies *P. a. aequinoctialis* which breeds on Atlantic and Indian Ocean islands and that none of the birds caught were from the subspecies *P. a. steadi* that breed on New Zealand's sub-Antarctic islands (Techow et al. 2016). A yearly bycatch of 227 birds (including recorded and extrapolated bycatch) within the South African pelagic longline fishery represents <0.1% of the overall population of *P. a. aequinoctialis*, and is unlikely to have a significant impact on the subspecies. However, the bycatch may contribute to the decline of the Crozet Island population which is already under pressure from the Patagonian toothfish fishery operating around the island (Barbraud et al. 2008).

Shy-type albatrosses, represented off South Africa by *T. steadi* (95%) and *T. cauta* (5%; Baker et al. 2007; Petersen et al. 2009a; this study), are the most commonly recorded albatross species in South African waters. The proportion of immature birds killed is similar to the proportion observed in South African waters (Hockey et al. 2005). There was a female bias in shy-type albatross bycatch,

but females were only present in immature birds. Adult female albatrosses tend to disperse further afield from breeding islands than adult males (Weimerskirch et al. 2005), thus the high levels of immature females recorded as bycatch might reflect sexual segregation of foraging zones among immatures; similarly many more female northern and southern royal albatrosses (Jiménez et al. 2016a) and wandering albatrosses (*D. exulans*; Jiménez et al. 2016b) are recorded as longline bycatch off South America. I estimate that 97 shy-type albatrosses are killed each year by the South African pelagic longline fishery, or five *T. cauta* and 92 *T. steadi*, roughly six times less than bycatch levels from 1998–2005 (Petersen et al. 2009a). The current bycatch estimates of shy-type albatrosses from the pelagic longline fishery operating off South Africa are unlikely to have significant effects on the global population (Baker et al. 2007), with < 0.01% of both populations killed annually for *T. cauta* and *T. steadi*, respectively (Aldermann et al. 2011; Baker et al. 2013).

Immature birds comprised the greatest proportion of black-browed albatrosses killed in this study. These immature birds are probably resident in South African waters for their first 3–4 years (Hockey et al. 2005) which could account for the higher levels of bycatch for this age group. Additionally, Barbraud et al. (2008) speculate that immature birds are at greater risk than adults as they are less efficient foragers and thus need to take greater risks behind fishing vessels, or are less experienced at foraging behind vessels without getting hooked. Breeding numbers have increased in recent years, particularly at the Falklands and islands off Chile, resulting in the species being down-listed to Near-Threatened (IUCN 2016). However, the population on South Georgia continues to decrease, apparently due to mortality from various fisheries (Croxall 2008; Poncet et al. 2017). There are an estimated 56 000 breeding pairs on South Georgia totalling around 200 000 birds (including immature and non-breeding birds; ACAP unpubl. data). The annual bycatch of c. 19 birds (also roughly six times lower than in 1998–2005; Petersen et al. 2009a) represents < 0.01% of the South Georgian population and is thus unlikely to be a major driver of this population's ongoing decline, however demersal longline fisheries locally (South Africa and Namibia) may also be contributing to their decline (Petersen et al. 2009b).

The Endangered Atlantic yellow-nosed albatross is endemic to Tristan da Cunha and Gough islands, and mainly occurs off the southwest and west coasts of southern Africa, avoiding the east coast (Hockey et al. 2005; ACAP 2009a). Atlantic yellow-nosed albatross is one of the most frequently caught species in pelagic longline operations off Brazil (Bugoni et al. 2008) and Namibia (Petersen et al. 2007), with smaller numbers off South Africa (Petersen et al. 2009a; this study). Based on confirmed proportions of the two yellow-nosed albatross species caught (20%; Table 2.2),

only c. 5 birds are estimated to be killed each year by pelagic longlining off South Africa, which in isolation is unlikely to have a severe impact on this species' global population.

Indian yellow-nosed albatrosses breed on four island groups in the southern Indian Ocean, but it is thought that only birds from PEI and Crozet Islands regularly visit South African waters (Weimerskirch et al. 1985). They have a similar global population (41 580 breeding pairs; IUCN 2016) to Atlantic yellow-nosed albatrosses (c. 40 000 breeding pairs; ACAP 2009b), and are listed as Endangered (IUCN 2016). Large numbers are killed by tuna longliners in subtropical waters (Weimerskirch and Jouventin 1998), off Australia (Gales et al. 1998) and South Africa (Petersen et al. 2009a). The c. 20 birds killed each year by longline fleets off South Africa is unlikely to significantly affect the global population, however combined bycatch from other fisheries may influence some populations.

Almost four times as many Cape gannets were killed by pelagic longline fisheries off South Africa during the study period, compared to between 1998 and 2005 (Petersen et al. 2009a). Cape gannets are restricted to just six breeding islands off southern Africa and their numbers have been decreasing steadily over the last 50 years (Crawford et al. 2007). Despite the only south coast breeding locality (Bird Island in Algoa Bay) now supporting > 50% of the world population (Crawford et al. 2007), larger numbers of birds were killed off the west and southwest coasts and Agulhas Bank, which are likely to have originated from the decreasing west coast colonies (Pichegru et al. 2007). With more limited foraging opportunities off the west coast, due to competition with fisheries (Cohen et al. 2014), gannets from that region are forced to scavenge from pelagic longline vessels more regularly than those from Bird Island (Moseley et al. 2012; Grémillet et al. 2016).

Diomedea albatrosses are noteworthy for the small numbers killed in my study, compared to studies off South America (Bugoni et al. 2008; Jiménez et al 2016 a; b), Australia (Gales et al. 1998) and New Zealand (Waugh et al. 2008). Petersen et al. (2009a) also did not record any wandering albatrosses in bycatch autopsies from the South African pelagic longline fishery from 1998–2005. The paucity of *Diomedea* albatross bycatch mortality off South Africa is likely explained by their relatively low abundance in South African waters; the ratio between *Diomedea* albatross and shy-type albatross attending pelagic longline vessels in offshore waters off southern Africa has been recorded as 1:8 (Chapter 3). However the bycatch ratio between these two species groups off South Africa is much larger (1:60) and thus it is likely there is another reason for the low bycatch. A theory put forward by Barbraud et al. (2013) is that there has been selection since the 1960s against wandering albatrosses that are susceptible to being killed at fishing vessels through differential mortality, thus the more susceptible individuals have already been killed as longline bycatch.

2.4.3 Factors affecting seabird bycatch

The higher bycatch rates by South African vessels compared to the Asian fleet is most likely related to the differences in target species, fishing operations and hence permit conditions between these fleets. Longer branchlines increase the risk of seabird bycatch due to their slower sink rates, which allows scavenging seabirds more time to access baited hooks (Petersen et al. 2009a; Melvin et al. 2013). During my study the South African fleet had higher bycatch rates than the Asian fleet, despite their use of shorter branchlines, however differences in branchline length between the two fleets is likely to be trivial compared to the large differences in bird-scaring line usage, line weighting and setting times.

As was predicted lunar luminance played an important role in explaining bycatch rates, probably because there is increased nocturnal seabird activity around full moon (Mackley et al. 2011) and moonlight allows birds to scavenge more effectively (Jiménez et al. 2009). Year was also an important variable affecting seabird bycatch numbers in the Asian fleet. It is likely that year was not identified as significant for South African vessels due to the much smaller sample size for this fleet, with no observer data for the final three years of the study (Figure 2.8).

Both area and season were important factors affecting seabird bycatch. This is to be expected, as many of the seabird species visiting South African waters are primarily seasonal migrants to South African waters. Most seabird species that breed in the Southern Hemisphere visit South African waters during their non-breeding seasons of winter and spring (Crawford et al. 1991), which is when bycatch off South Africa was highest. Larger numbers of seabirds are found in the more productive waters off Cape Point, the west coast and the Agulhas Bank (Crawford et al. 1991). These areas showed substantially higher rates of seabird bycatch than the east coast and international waters, where seabird numbers are lower (Crawford et al. 1991; Chapter 3).

Bird-scaring lines were used on 95% of sets and reduced seabird bycatch for the Asian fleet, compared to sets without bird-scaring lines. Bird-scaring lines did not negatively affect seabird bycatch on South African-flagged vessels, probably because bird-scaring lines are deployed more regularly when fishing in areas of high seabird abundance;76% in high-bycatch areas, compared to 62% bird-scaring line compliance for sets in low-bycatch areas. The use of bird-scaring lines on 95% of observed sets during my study represented a marked improvement from 1998–2005, when they were used on 51% of sets (Petersen et al. 2009a). Bird-scaring lines have proven particularly effective at reducing seabird bycatch in pelagic longline fisheries (Petersen et al. 2009a; Melvin et al. 2013).

Wind strength affected the bycatch of yellow-nosed albatrosses only, but surprisingly bycatch decreased with increasing wind strength (although wind direction was not considered). Bird-scaring lines become less effective when high winds blow them sideways, allowing birds to settle behind the vessel and access baited hooks. Melvin et al. (2013) found that most birds were caught during the lowest and highest Beaufort values (1 and 6–7, respectively). Bird-scaring line efficacy is perhaps reduced during periods of little or no wind, as streamers are no longer blown around unpredictably and thus do not protect the area behind the vessel from scavenging seabirds.

2.4.4 Management recommendations

Despite a number of stipulations pertaining to seabird bycatch within South African pelagic longline permit conditions, rates still remain higher than the South African target of 0.05 birds killed per 1 000 hooks (DEAT 2008). However bycatch rates for the Asian fleet have been close to the national target since 2010, with significant reductions in the last four years of the study (2010–2013). Unfortunately there are no observer data for the South African fleet in recent years (2011–2013). Better compliance with the use of seabird bycatch mitigation measures in the Asian fleet and improved seabird bycatch legislation are almost certainly the reasons for the reduced seabird bycatch rates.

Despite improved permit conditions, a number of sections could be further improved. The only reference to moon phase in the permit conditions is that once vessels have reached a limit of 25 seabird mortalities, when fishing on the high seas, these vessels are prohibited from fishing for three days around full moon (DAFF 2013). Given the strong influence of lunar luminance on seabird bycatch, I suggest that permit conditions should require additional mitigation measures over periods of high lunar luminance as a matter of course, not just after catching > 25 birds. Such measures could include a second bird-scaring line, adding more weight to lines or moving weights closer to the hook.

Bird-scaring lines were used to good effect by Asian vessels for most sets during the study period. However, observers reported they were rarely used by South African vessels, and are likely to be used even less on unobserved trips. In addition, some bird-scaring lines are incorrectly deployed, or have broken or missing streamers, thus providing inadequate protection. Regularly checking and maintaining bird-scaring lines would ensure better compliance with permit requirements (DAFF 2013).

Despite time of setting not proving a significant variable affecting seabird bycatch in the South African fleet, it has influenced seabird bycatch in many other studies (Gales et al. 1998; Petersen et

al. 2009a; Trebilco et al. 2010) and the effectiveness of night-time setting in reducing seabird bycatch in the pelagic longline fishery has been well documented (Brothers et al. 1999). In my study seabird bycatch was slightly higher around sunrise, although the difference was not significant, this does which suggest that vessels should ensure all line setting is concluded well before sunrise, when seabird activity is known to be high (Ainley et al. 1984; Barnes et al. 1997). I thus suggest that the exemption allowing South African vessels to set lines during daylight hours should be amended, unless vessels can demonstrate a seabird bycatch rate of below 0.05 birds per 1 000 hooks.

The low observer coverage throughout the study period along with the complete lack of observer coverage from 2011-2013 on South African vessels impacts the accuracy of estimates of seabird bycatch figures from this fleet and thus should be greatly improved. Seabird bycatch rates are expected to be higher on unobserved trips than observed trips, as compliance with seabird mitigation measures is likely to be lower on unobserved trips. Observer programmes for a number of locally relevant RFMOs require a minimum of 5–20% observer coverage (ICCAT 2014; IOTC 2014). This figure should be higher when fishing in areas of high seabird abundance, south of 25^{o} S. Another option to ensure compliance with mitigation measures is electronic monitoring (via video cameras) of various fleets. Piasente et al. (2012) concluded that electronic monitoring was an effective alternative to onboard observers in the eastern tuna and billfish fishery off Australia; seabird bycatch, mitigation measure compliance and seabird abundance data were all comparable to onboard observer data.

Although seabird bycatch rates have decreased from the previous longline bycatch study of the pelagic fishery off South Africa (Petersen et al. 2009a) and especially since 2008 (when significant changes to permit regulations were made), the rate in recent years has seldom met the target rate of 0.05 birds per 1 000 hooks stipulated in South Africa's NPOA-Seabirds (DEAT 2008). As a result, unacceptably large numbers of seabirds continue to be killed each year in South Africa's waters, including species of conservation concern. Better compliance with existing seabird bycatch regulations should further reduce seabird bycatch. Additional study of the factors affecting seabird bycatch should further reduce bycatch by improving and strengthening regulations.

CHAPTER 3

SPATIAL VARIATION IN SEABIRD ASSEMBLAGES ASSOCIATED WITH PELAGIC LONGLINE VESSELS OFF SOUTHERN AFRICA: IMPLICATIONS FOR BYCATCH

Abstract

Data collected by fisheries observers onboard pelagic longline vessels off South Africa have provided a good understanding of the numbers and composition of seabird bycatch in this fishery. However, little is known about the structure of the seabird assemblage associated with these vessels, how it differs throughout the region, and how this affects bycatch risk. Seabird counts were conducted from two Asian pelagic longline vessels in winter and early spring, the time of peak fishing activity, in three regions off southern Africa: along the shelf edge of the Agulhas Bank off the South African south coast, oceanic waters off the South African east coast, and off southern/central Mozambique. Daily counts were conducted while fishing on 159 days, recording 39 seabird taxa groups. The greatest numbers of seabirds were recorded off the Agulhas Bank, where white-chinned petrels were the most abundant species, and the lowest numbers of seabirds were recorded off Mozambique. Species richness was greatest off the Agulhas Bank and Mozambique coast. Many seabird species associated with the vessels have not been recorded as pelagic longline bycatch off South Africa, but most of the commonly recorded bycatch species were observed in large numbers. The large numbers of white-chinned petrels and shy-type albatrosses observed at vessels operating off the Agulhas Bank and east coast, explain their high levels of pelagic longline bycatch off South Africa. However, for some albatross species (e.g. shy, black-browed and wandering-type *D. exulans*/*dabbenena*) there was a mismatch between abundance and bycatch rates, suggesting that factors other than abundance drive the bycatch of these species. Differences in foraging techniques may mean certain species are more susceptible to seabird bycatch than other species; such information is important for the implementation of seabird conservation plans, and emphasizes the need to explore differences and similarities in bycatch of global longline fisheries.

3.1 Introduction

Commercial fishing operations have large direct and indirect impacts on many seabird species throughout the world's oceans (Furness 2003). Most of these impacts are negative, but some fisheries have benefitted certain seabird populations. One of the most severe threats impacting seabird populations is seabird mortality from fisheries bycatch (Croxall et al. 2012). Seabird bycatch occurs in many different types of commercial fisheries (Perez and Wahrlich 2005; Bull 2007; Anderson et al. 2011), with trawl, driftnet and longline fisheries considered the most detrimental (Croxall 1998).

Seabirds are also negatively affected by the depletion of fish stocks and the resultant reduction in food availability (Lewison et al. 2012). Conversely, other seabird species have benefitted from increased food availability from fisheries, which would not naturally be available to foraging seabirds, such as offal and discards of benthic fish species (Furness 2003). The provision of food from fishing vessels has altered some seabird species' distributions (Votier et al. 2004; Bartumeus et al. 2010). The expansion of commercial fisheries was a significant driver for the expansion of northern fulmars (*Fulmarus glacialis*) across Europe (Thompson 2006), and the distributions of several seabird species is influenced by commercial trawl fisheries in the southern Benguela region off South Africa (Ryan and Moloney 1988). However, attraction of seabirds to fishing vessels may also increase their risk of accidental mortality on fishing gear (Watkins et al. 2008; Jiménez et al. 2011).

The productive waters off South Africa support a number of different fisheries; the deep-water hake trawl fishery is the most commercially valuable fishery, while demersal and pelagic longlining also are important fisheries (FAO 2006). Seabird bycatch has decreased in recent years in all three of these fisheries (Petersen et al 2009a; Maree et al. 2014; Chapter 2), but it remains above the national interim target rate of less than 0.05 birds killed per 1000 hooks set within the pelagic longline fishery (Chapter 2). An estimated 450 birds are killed each year by pelagic longline fleets operating off South Africa, targeting swordfish and various tunas. Bycatch is mostly comprised of white-chinned petrels (c. 66%) and *Thalassarche* albatrosses (c. 31%), with smaller numbers of gannets and other species (Chapter 2).

Despite the relatively high levels of seabird bycatch within this fleet, there is little information on seabird assemblages associated with pelagic longline vessels off southern Africa. Fisheries observers are required to collect seabird assemblage data, however these data have not been analysed and are likely to be inaccurate for a number of difficult-to-identify species groups. At-sea observations of seabirds while onboard research and fishing vessels in South African waters have produced a good understanding of the distribution of most seabirds (Ryan 2017), however there are very few such at-sea observations from Mozambique waters (Ryan et al. 2006). A number of studies quantifying seabird assemblages around both demersal (Olmos 1997; Gandini and Pon 2007) and pelagic longline vessels (Jiménez et al. 2011) have been undertaken off South America. Jiménez et al. (2011) found that species composition and abundance rather than species richness influenced seabird bycatch rates. Apart from Jiménez et al. (2011), very little research has been conducted on linking seabird attendance rates to longline bycatch.

I describe seabird assemblages associated with pelagic longline vessels off southern Africa based on the species richness and abundance of seabirds attending two Asian pelagic longline vessels in three

fishing areas off the south and east coasts. I predict that seabird richness and abundance is highest in the cooler, more productive waters of the Agulhas Bank. I also investigate any variation in the size or composition of the seabird assemblage throughout the hauling process to test whether seabird abundance increases as hauling progresses. By relating seabird attendance to seabird bycatch my study helps to better understand the spatial pattern and species composition of seabird bycatch associated with longliners off southern Africa. I hypothesise that seabird species considered particularly susceptible to pelagic longline bycatch will be some of the most abundant species associating with these vessels in my study.

3.2 Methods

3.2.1 Study area

Fishing effort was distributed in three regions; along the edge of the Agulhas Bank off the south coast between Cape Agulhas and Port Alfred (Agulhas Bank), in oceanic waters outside the South African 200 nm EEZ off the east coast and in the Mozambique Channel off southern and central Mozambique (Figure 3.1). It was decided to combine the fishing areas off Cape Agulhas and Port Alfred as one region (Agulhas Bank), rather than two separate regions (Chapter 2) as sample sizes would have been further reduced and these areas were relatively close apart with similar sea surface temperatures (SST). All fishing effort off the Agulhas Bank was within the South African EEZ, and all fishing off the Mozambique coast was within the Mozambique EEZ. The Agulhas Bank and east coast region are characterised by cool water temperatures in winter which are strongly influenced by the Agulhas Current (Demarcq et al. 2003), while the Mozambique coast that lies farther north than the two other regions, receives warmer, tropical water from the Mozambique Current (Quartly & Srokosz 2004).

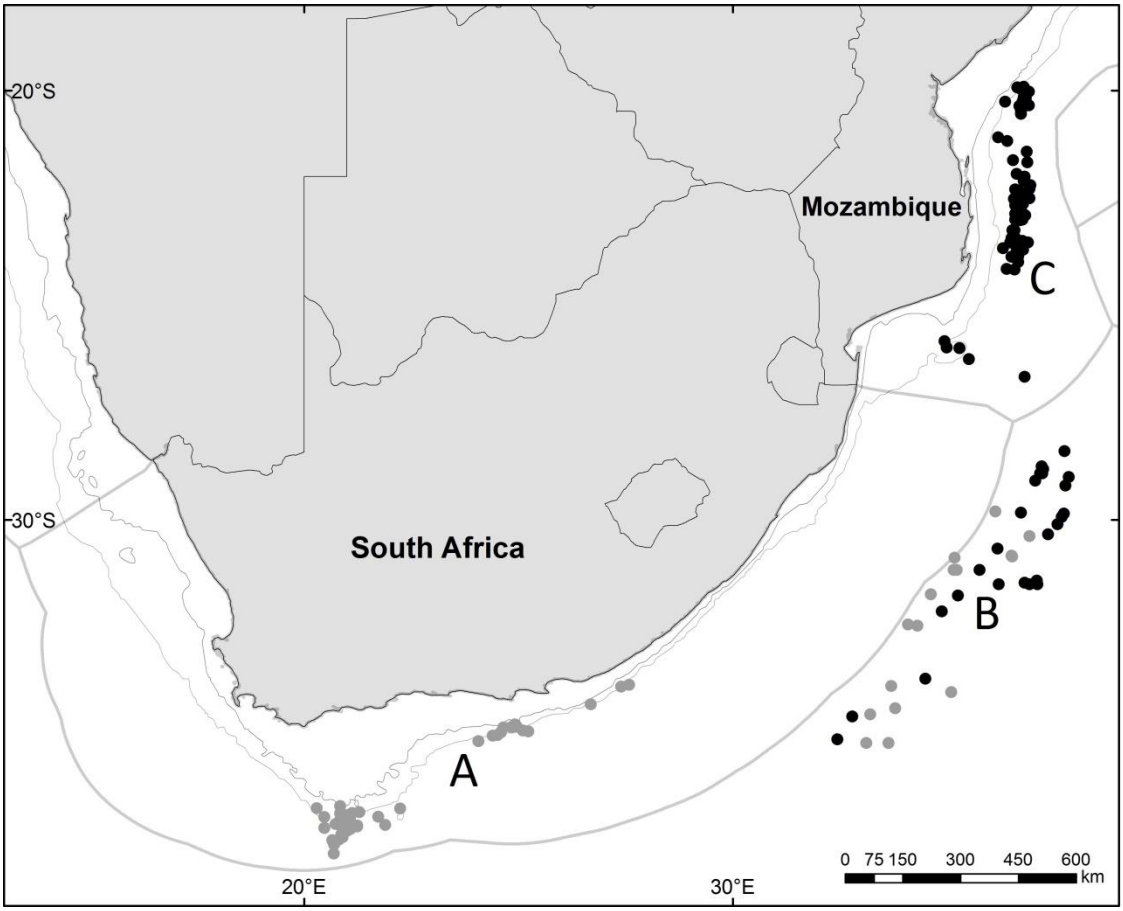


Fig. 3.1. Distribution of fishing effort off southern Africa during the two research trips in 2013 (grey dots) and 2015 (black dots), in three fishing regions Agulhas Bank (A), east coast (B) and Mozambique Channel (C).

3.2.2 Fishery characteristics and seabird bycatch

Pelagic longlining off South Africa is comprised of two fleets; a domestic fleet, which mostly targets swordfish and a foreign-flagged (Asian) fleet, which targets tunas (Chapter 2). Fishing effort for Asian vessels peaks in winter and spring (April–October) with fishing effort dispersed throughout the year for the local South African fleet. Approximately 4.4 million hooks are set each year by the combined fleets (Chapter 2). These data include Asian vessels holding South African joint venture fishing permits and fished in international waters adjacent to South Africa. Both local and foreign pelagic longline fleets, targeting various tuna species (especially yellowfin tuna), operate in Mozambican waters year-round (Ternon et al. 2014; Chacate and Mutombene 2015).

Seabird bycatch levels from pelagic longliners off South Africa have previously been considered some of the highest in the world (Anderson et al. 2011), but in recent years seabird bycatch has been reduced (Chapter 2). Moderate levels of seabird bycatch occurred off the Cape Point, Agulhas Bank and the south coast region, with lower levels recorded in international waters off the east coast (Chapter 2). Seabird bycatch in Mozambican waters is not as well documented, but no bycatch was observed in the pelagic longline fishery in 2014 and 2015 (Chacate and Mutombene 2015; Mutombene 2015), including the fishing effort observed in this study. Bycatch susceptibility of a species was considered 'high' if > 10 birds are killed per year by pelagic longlines off South Africa (Chapter 2), 'moderate' if 1 – 10 birds are killed per year, 'low' if the species has not been recorded as pelagic longline bycatch off South Africa but has been killed by pelagic longline fisheries elsewhere in the world, and 'no risk' if the species has never been recorded as pelagic longline bycatch by any fleet worldwide. Seabird bycatch figures were extracted for the Agulhas Bank and the international waters off the east coast from the national fisheries observer bycatch records.

3.2.3 Description of fishing operations

Research was conducted onboard two Asian pelagic longline vessels (labelled A and B) which fished from July–September 2013 (A) and July–October 2015 (B). Vessels were both c. 49 m in length and targeted tunas and swordfish. Vessel A fished off the Agulhas Bank and east coast while Vessel B fished off the east coast and in the Mozambique Channel (Table 3.1). Fishing off the Agulhas Bank was conducted between late-July and mid-September, fishing off the east coast in July only and fishing in the Mozambique Channel from August to mid-October (Table 3.1). While fishing along the east coast Vessel A commenced line setting in the early morning (05h00–06h30) with hauling starting in the early afternoon (14h00–16h00), while off the Agulhas Bank line setting commenced at night (22h30–02h30) and hauling in the early morning (06h00–08h30). Vessel B commenced line setting in the early morning (03h30–07h00) and line hauling in the afternoon (14h00–16h30), irrespective of fishing regions. Vessel A and B averaged c. 2700 and 3100 hooks per set, respectively. SST was recorded at the beginning of each set. During hauling, both vessels discarded bait and offal periodically on the starboard side of the vessel, the same side where line hauling took place.

Table 3.1. Summary of fishing sets and number of days during which seabird counts were performed in the three different fishing regions.

Vessel	Agulhas Bank		East coast		Mozambique Channel	
	Fishing sets (months)	Counts (days)	Fishing sets (months)	Counts (days)	Fishing sets (months)	Counts (days)
A	45 (Jul–Sep)	41	17 (Jul)	16	-	-
B	-	-	28 (Jul)	26	76 (Aug–Oct)	76
Total	45	41	45	42	76	76

3.2.4 Data collection

Counts were performed (daylight hours during the line haul only) from the stern of the vessel in a 180^{o} degree arc up to 300 m behind the vessel. Counts were only performed during day-time hauling operations because setting operations were mostly conducted during the night when visibility was much reduced, making seabird counts less accurate. In areas of high seabird abundance, estimates of species abundance were made, rather than absolute counts. While onboard Vessel A, I aimed to perform seabird counts at the start of hauling and then at hourly intervals throughout the entire haul, however due to observer duties and adverse weather, this was not always possible. To obtain daily species totals for each species the maximum number of birds recorded in a single count was used. Due to low seabird abundance attending Vessel B, hourly counts were not performed; instead maximum counts were recorded for each species throughout the day's fishing. Due to difficulties of at-sea identification, a number of species were not identified to species level but rather lumped into species groups. Wandering and Tristan albatross (*D. dabbenena*) were lumped as wandering-type albatrosses, shy and white-capped albatross as 'shy-type albatrosses', northern and southern giant petrels as 'giant petrels', while prions (*Pachyptila*) were not identified to species and lumped as 'prions' (although Antarctic prion *P. desolata* likely dominated throughout). Indian and Atlantic yellow-nosed albatross were occasionally lumped as 'yellow-nosed albatross sp.' when confronted with large numbers of albatrosses whilst fishing off the Agulhas Bank (a zone of overlap) only. Only birds seen associating with the vessel were included in counts; this meant a number of species were seen but not counted. The most abundant species which was seen but excluded from counts were sooty terns (*Onychoprion fuscatus*) which were regular in the Mozambique Channel. This species was generally seen in large flocks of up to several hundred birds, but they did not associate with the vessel or consume vessel discards.

3.2.5 Data analyses

I used canonical correspondence analysis to explore the spatial variation in the composition and abundance of the assemblage of seabirds attending the fishing vessels. All (*n* = 149) daily seabird totals were incorporated into this analysis. To assess possible differences in species richness between the three regions, I performed sample-based rarefaction analyses. Both analyses used the package 'vegan' in the R statistical package (R Core Team 2015). To test for differences in total abundance of seabirds between the three different regions I used Poisson regression within the R statistical package, to test the interaction between the two fixed effects, 'region' and 'year' (Agresti 2002). For each species recorded in the three different regions, the total number, mean, standard deviation, daily maximum and % frequency occurrence (FO; number of days on which the species was recorded as a percentage of total days) were all recorded.

To investigate the effect of time after commencement of hauling on the size of the seabird assemblage associated with the vessel (Vessel A only), generalised linear mixed models in the package 'lme4' within the R statistical package were used (Bates et al. 2008). Models were created testing hourly counts as a discrete response variable, investigating hourly differences in seabird abundance throughout the day. Models were also created to test whether seabird abundance at the start of hauling differed from all other counts throughout the remainder of the haul. To account for the variation in the seabird assemblage according to fishing day, 'hours after haul commencement' was nested within 'day number', with 'region' also included as a factor. Values given are means ± standard deviation, unless otherwise stated.

3.3 Results

Seabird counts were conducted during 159 hauls, recording 20 557 birds from 39 taxa associating with the two vessels. The length of time spent fishing in each region (Table 3.1) played a large role in the species richness for the different regions, but species accumulation curves (Figure 3.2) suggest that species richness had plateaued for both the Agulhas Bank and east coast but was still increasing slightly for the Mozambique Channel. Average SST was similar along the edge of the Agulhas Bank (19.9 ± 1.4^{o} C) and east coast (20.5 ± 1.0^{o} C), but much warmer in the Mozambique Channel (24.9 ± 0.8^{o} C). While fishing off the east coast from Vessel A in 2013, SST was slightly cooler (19.8 ± 0.7^{o} C) than when fishing off the east coast from Vessel B in 2015 (20.9 ± 1.0 ^{o} C).

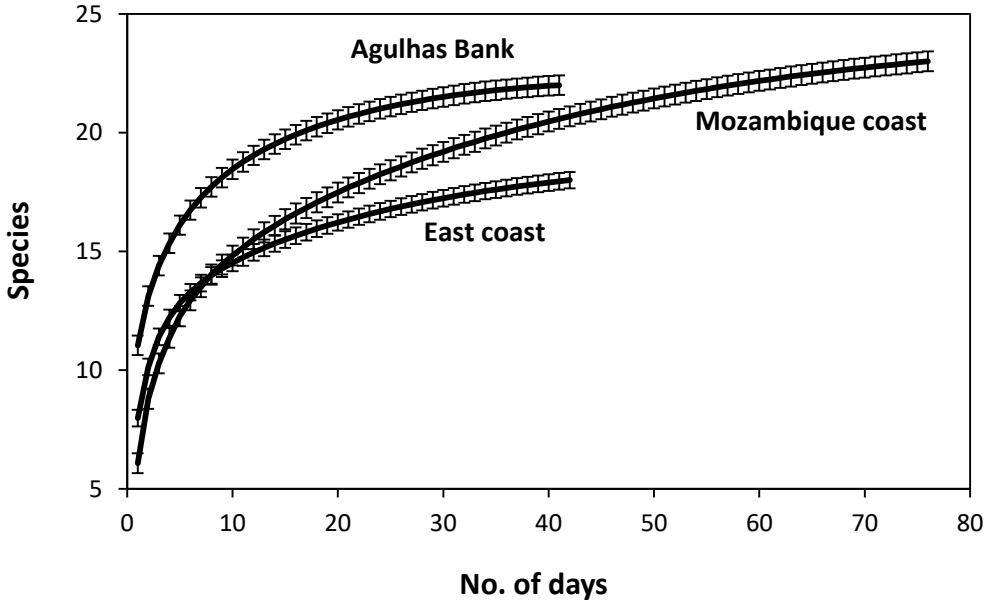


Fig. 3.2. Species accumulation (rarefaction) curves for the three fishing regions. Error bars represent ± 1 SE.

3.3.1 Spatial and temporal variation in seabird assemblage

Seabird abundance and composition varied significantly between the three different fishing regions (Figures 3.3 and 3.4; Appendices 3.1 – 3.3). Both 'region' ($p < 0.001$) and 'year' ($p < 0.001$) significantly affected total daily abundance. As expected, seabird abundance was greatest off the Agulhas Bank (Figure 3.5), with species richness was similar there and in the Mozambique Channel. Daily seabird totals averaged 281 ± 130, 37 ± 56 and 22 ± 56 birds per day for the Agulhas Bank, east coast and Mozambique Channel, representing 21, 18 and 22 seabird taxa, respectively (Table 3.2). White-chinned petrels were the most frequently recorded species off the Agulhas Bank (100% FO) as well as off the east coast (76% FO), while great frigatebirds (*Fregata minor*; 74% FO) were the most frequently recorded species in the Mozambique Channel. Sooty terns were the most abundant species off the Mozambique coast, but they did not attend the vessel. A number of species were only recorded in one of the three fishing regions (Table 3.2). The Agulhas Bank shared 50 and 43% of taxa recorded off the east and Mozambique coasts, respectively, while the east coast and Mozambique Channel shared 48% of taxa. Almost all seabird taxa off the Agulhas Bank (91%) and east coast (89%) were Procellariiformes, but species from this order

were less dominant in the Mozambique Channel (73%). The east coast was the only region in which counts from both vessels were conducted (Table 3.1); despite no obvious differences in discard practices between vessels, seabird abundance was significantly higher (t = 2.11; p = 0.002) from Vessel A (average 88 ± 63 birds per day) compared to Vessel B (average 29 ± 22 birds per day) whilst fishing off the east coast, although it must be noted that counts from the two vessels were conducted two years apart.

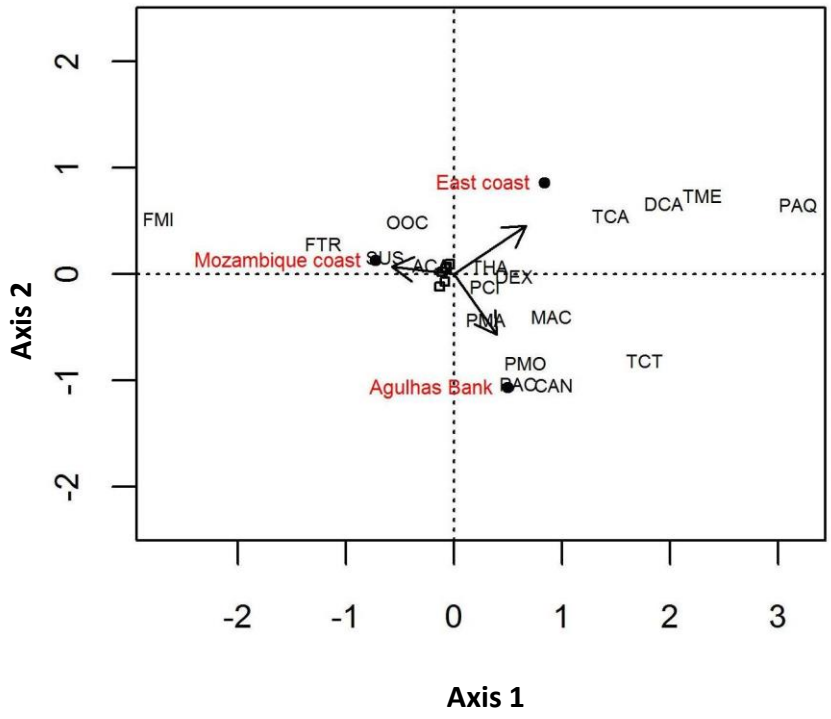


Fig. 3.3. Canonical correspondence analysis of the spatial variation in abundance of seabirds attending pelagic longline vessels off southern Africa. Only species with > five individuals observed were included in the analysis. Species codes as in Table 3.2. ☐ = overlapping species (CMA, DEP, DSA, MCA, PGR, SPA, TCH).

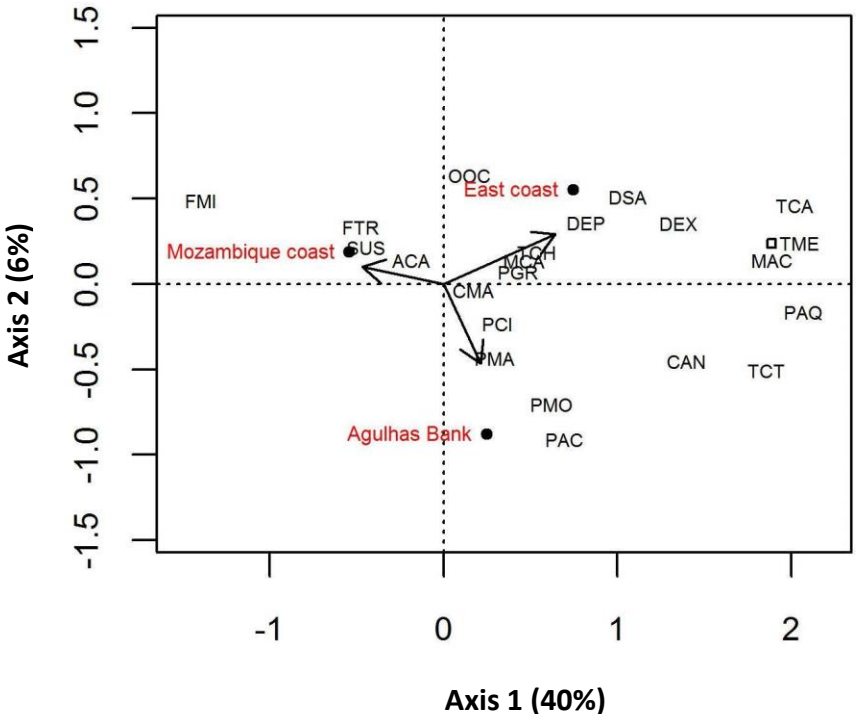


Fig. 3.4. Canonical correspondence analysis of the spatial variation in presence of seabirds attending pelagic longline vessels off southern Africa. Species codes as in Table 3.2. ☐ = overlapping species (DCA).

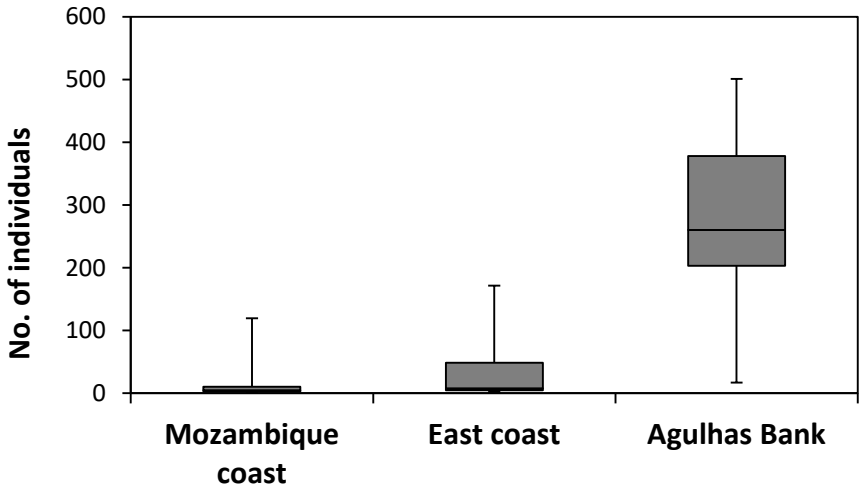


Fig. 3.5. Spatial variation in in seabird assemblage size between the Mozambique Channel (*n* = 76), east coast (*n* = 42) and Agulhas Bank (*n* = 41). Whisker plots represent the 95% confidence intervals, grey boxes represent the 25th and 75th percentile while the black line represents the median.

Table 3.2. Seabird species observed associating with pelagic longline vessels, summarised by fishing region; Agulhas Bank (*n* = 41 d), east coast (*n* = 42 d) and Mozambique Channel (*n* = 76 d). Species are listed in descending order of % bycatch composition off South Africa. %CO refers to the total composition of seabirds attending the vessel per region, while %FO (frequency occurrence) refers to the number of days on which the species was recorded as a percentage of total days. See methods for 'Bycatch susceptibility' definition.

Species	Bycatch susceptibility	Species code	Agulhas Bank			East coast			Mozambique Channel		
			Daily mean	%CO	%FO	Daily mean	%CO	%FO	Daily mean	%CO	%FO
White-chinned petrel	High	PAQ	114	40	100	12	31	76	1	6	12
Shy-type albatross	High	TCA	22	8	98	2	4	33	< 1	< 1	1
Black-browed albatross	High	TME	65	23	98	6	17	48	< 1	< 1	4
Indian yellow-nosed albatross	High	TCT	23	8	88	6	17	90	1	6	16
Atlantic yellow-nosed albatross	High	TCH	< 1	< 1	17	-	-	-	-	-	-
Yellow-nosed albatross spp.	High	THA	2	< 1	10	-	-	-	-	-	-
Cape gannet	Moderate	MCA	< 1	< 1	12	-	-	-	-	-	-
Giant petrel sp.	Moderate	MAC	3	< 1	88	< 1	2	43	< 1	< 1	4
Wandering/Tristan albatross	Moderate	DEX	2	< 1	61	< 1	1	17	-	-	-
Northern royal albatross	Moderate	DSA	< 1	< 1	49	-	-	-	-	-	-
Southern royal albatross	Moderate	DEP	< 1	< 1	34	-	-	-	-	-	-
Brown Skua	Moderate	CAN	1	< 1	56	1	3	64	< 1	2	7
Grey petrel	Moderate	PCI	-	-	-	< 1	1	14	-	-	-
Cape petrel	Moderate	DCA	40	14	100	4	11	48	< 1	< 1	1
Salvin's albatross	Low	TSA	< 1	< 1	1	-	-	-	-	-	-
Grey-headed albatross	Low	TCR	-	-	-	< 1	< 1	2	-	-	-
Sooty albatross	Low	PFU	-	-	-	< 1	< 1	7	-	-	-
Light-mantled albatross	Low	PPA	-	-	-	< 1	< 1	5	-	-	-
Spectacled petrel	Low	PCO	< 1	< 1	5	-	-	-	-	-	-
Great-winged petrel	Low	PMA	< 1	< 1	5	< 1	< 1	33	< 1	2	11
Flesh-footed shearwater	Low	ACA	-	-	-	-	-	-	6	28	20
Sooty shearwater	Low	PGR	< 1	< 1	5	< 1	< 1	5	< 1	< 1	3
Red-footed booby	Low	SUS	-	-	-	-	-	-	< 1	2	32
Great frigatebird	Low	FMI	-	-	-	-	-	-	3	15	74
Soft-plumaged petrel	No risk	PMO	< 1	< 1	10	< 1	2	50	< 1	< 1	4
Blue petrel	No risk	HCA	< 1	< 1	12	-	-	-	-	-	-
Prion sp.	No risk	PAC	< 1	< 1	15	4	10	67	3	12	8
Wilson's storm-petrel	No risk	OOC	< 1	3	7	< 1	< 1	2	2	9	37
Black-bellied storm-petrel	No risk	FTR	7	< 1	41	-	-	-	3	15	38
Swinhoe's storm-petrel	No risk	HMO	-	-	-	-	-	-	< 1	< 1	3
Matsudaira's storm-petrel	No risk	HMA	-	-	-	-	-	-	< 1	< 1	3
White-faced storm-petrel	No risk	PMA	-	-	-	-	-	-	< 1	< 1	1
South polar skua	No risk	CMA	-	-	-	< 1	< 1	5	< 1	< 1	5
Kelp gull	No risk	LDO	-	-	-	-	-	-	< 1	< 1	1
Arctic tern	No risk	SPA	-	-	-	-	-	-	< 1	2	4

Most counts (55%) were performed within three hours after the commencement of hauling (Fig 3.6). Despite the total number of seabirds averaging smallest at the start of hauling (Figure 3.6), seabird numbers associated with Vessel A were not affected by 'hours after haul commencement' (SE ± 0.083; p = 0.303). Species richness was not affected by 'hours after haul commencement' (SE ± 0.009; p = 0.419). The only species which was affected by 'hours after haul commencement' was the black-browed albatross, which gradually increased in number as the haul progressed (SE ± 0.012, p = 0.031). Seabird abundance at the start of hauling (112.6 ± 111.9) was lower than throughout the remainder of the haul (147.9 ± 88.7; SE ± 0.092; 95% confidence intervals 0.281 – 2.261; p < 0.001).

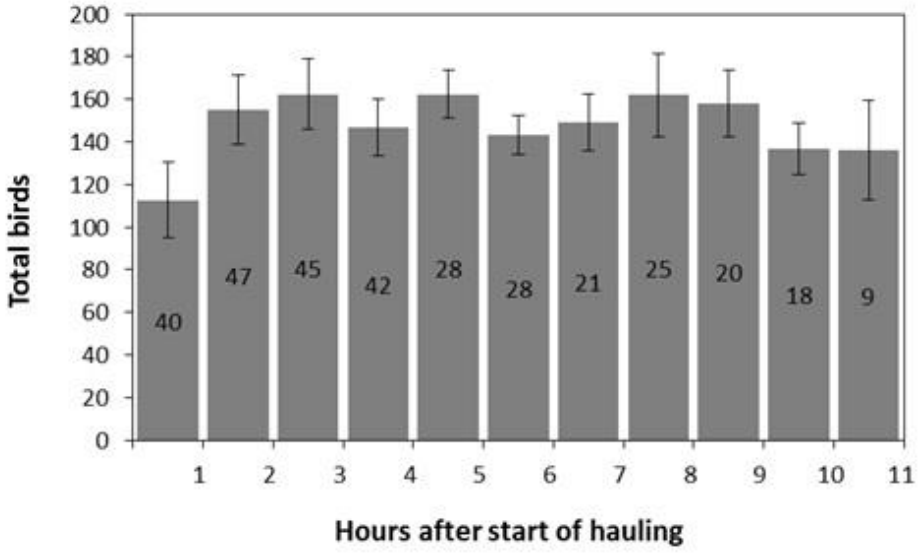


Fig 3.6. Variation in mean assemblage size of birds associating with Vessel A, grouped by number of hours after commencement of hauling. Numbers within the columns represent number of counts, with error bars representing ± SE.

3.3.2 Seabird bycatch

Large numbers of seabirds considered highly susceptible to pelagic longline bycatch were recorded associating with Vessel A off the Agulhas Bank, with smaller numbers of highly susceptible species observed off the east coast and in the Mozambique Channel (Table 3.2). Species considered highly or moderately susceptible to pelagic longline bycatch represented 97% of the assemblage off the Agulhas Bank; this figure was similar off the east coast (83%; Table 3.2) and lower in the Mozambique Channel (29%; Table 3.2). Almost half of the species attending the vessel in the Mozambique Channel are not

susceptible to pelagic longline bycatch (Table 3.2). Many of the species considered highly susceptible to longline bycatch were recorded in large numbers and with high %FO off the Agulhas Bank, however in very low numbers and %FO in the Mozambique Channel (Table 3.2). The five most abundant species recorded in the Mozambique Channel had low or no pelagic longline bycatch susceptibility, while the four and three most abundant species off the Agulhas Bank and east coast, respectively, had medium or high pelagic longline bycatch susceptibility.

The only seabird bycatch mortalities (*n* = 18) were recorded on Vessel A while fishing off the Agulhas Bank. Three species were killed: 15 white-chinned petrels, 2 shy-type albatrosses and 1 Indian yellow-nosed albatross. All of these birds probably were hooked during setting operations; an additional Indian yellow-nosed albatross hooked during hauling was released with minor injuries. Most birds (83%) were killed within 3 days of full moon, while using the full suite of required mitigation measures (night-time setting, two bird-scaring lines and weighted branchlines).

3.4 Discussion

3.4.1 Spatial variation in seabird assemblage

As was predicted, the highest seabird abundance occurred off the Agulhas Bank, with lowest abundance in the Mozambique Channel. My findings are similar to those of Ryan et al. (2002) who also recorded larger numbers of seabirds associating with pelagic longline vessels off the south coast (Agulhas Bank) compared to the east coast, west coast or adjacent international waters. The southern Benguela/Agulhas Bank region is one of the most productive oceanic regions on earth (Shannon and Fields 1985), resulting in high seabird abundances (Crawford et al. 1991). By comparison, the warmer tropical waters of the Mozambique and Agulhas Currents are less productive resulting in lower seabird abundance (Jaquemet et al. 2004). Tropical seabirds are more reliant on mesoscale eddies in these oligotrophic waters than seabirds in upwelling regions or at higher latitudes (Weimerskirch et al. 2004; Wanless 2015). Seabird numbers off the Agulhas Bank are likely somewhat artificially elevated by the presence of a number of fishing vessels, particularly deep-water hake trawlers, which attract large numbers of seabirds (sometimes in the thousands per vessel; Ryan and Moloney 1988). There are far fewer fishing vessels operating off Mozambique (Chacate and Mutombene 2015) and thus few seabirds probably are attracted to these waters to scavenge at vessels.

Species richness was similarly high whilst fishing in the Mozambique Channel and Agulhas Bank but lowest off the east coast. I had predicted species richness to be high from the Agulhas Bank, but such high richness off Mozambique was unexpected. Areas south of Africa, such as the Agulhas Bank, support a high diversity of procellariiform species, however the Mozambique Channel is not a significant area of species richness for procellariiform seabirds (Chown et al. 1998). Despite the lower procellariiform species richness at lower latitudes (Chown et al. 1998), seabird species richness is bolstered by seabirds from other orders. Almost one third of the species making up the seabird assemblage in the Mozambique Channel were species other than Procellariiformes, a much greater proportion than off the Agulhas Bank and east coast. Several species attending vessels in the Mozambique Channel breed on Europa Island (e.g. great frigatebird, red-footed booby *Sula sula*, etc.), and mainly forage within the Mozambique Channel (Le Corre et al. 2012).

The high degree of species overlap between the Agulhas Bank and east coast can be explained by a large number of fishing sets from both regions conducted at similar latitudes and SST. The least amount of overlap in seabird species associating with fishing vessels occurred between the Agulhas Bank and Mozambique Channel. The large difference in latitudes and water temperatures between the two regions is likely to have resulted in a different composition of seabird species.

Of the species only observed in the temperate waters of the Agulhas Bank, northern and southern royal albatrosses, are known to venture further north in winter, but are unknown from the east coast or Mozambique Channel (Hockey et al. 2005). Atlantic yellow-nosed albatross and spectacled petrel (*P. conspicillata*) are both species that are usually associated with the Atlantic Ocean and are rarely recorded off the east coast and Mozambique Channel (Hockey et al. 2005). Cape gannets cover large distances up the east coast following the sardine run in winter, but they generally remain close inshore, and within c. 500 km of their easternmost breeding colony at Bird Island in Algoa Bay (Hockey et al. 2005).

Fewer species were restricted to the east coast only, all of which were sub-Antarctic breeding species from the southwest Indian Ocean. Of these species, grey petrel was the only species regularly recorded (14% FO); all other species were seldom recorded (≤ 7% FO) and are rarely recorded from southern African waters (Hockey et al. 2005), but tracking studies show that sooty albatrosses (*Phoebetria fusca*) breeding on Marion Island frequently visit the east coast region (Schoombie et al. 2017). The east coast was the only region that did not produce sightings of black-bellied storm-petrels (*Fregetta tropica*),

probably because all fishing in this region took place in July, when this species is largely found in tropical waters; it typically occurs in southern African waters on passage in spring and autumn (Hockey et al. 2005). The reason for the large difference in seabird abundance between Vessel A and B while fishing off the east coast is unknown; both vessels had similar fishing techniques and discard practices, with similar quantities of fish caught, and thus neither of these factors should impact seabird abundance. Counts were made two years apart with cooler SSTs experienced while fishing from Vessel A, and thus the difference could be due to inter-annual variation in weather or oceanographic conditions, with the cooler SSTs perhaps attracting larger numbers of sub-Antarctic species to this region.

Almost all seabird species that were only recorded in the Mozambique Channel were species which generally only occur in tropical waters. Flesh-footed shearwater (*Ardenna carneipes*), the most abundant species off the Mozambique coast, is more common off the Mozambique coast than farther south (Hockey et al. 2005). Both red-footed booby and great frigatebird breed on Europa Island (Le Corre 2001), while Swinhoe's (*Oceanodroma monorhis*) and Matsudaira's storm-petrels (*Oceanodroma matsudairae*) are considered tropical seabird species (Onley and Schofield 2007). Surprisingly kelp gull (*Larus dominicanus*) a species regularly occurring in temperate waters, was only recorded in the Mozambique Channel, possibly because fishing was fairly close to the coast, compared to most fishing sets from other regions.

3.4.2 Hourly variation in seabird assemblage

Previous studies concluded that the size of the seabird assemblage at fishing vessels is influenced by fishing activity (Weimerskirch et al. 2000; Jiménez et al. 2011). Jiménez et al. (2011) found that seabird abundance around a pelagic longline vessel was greatest during hauling compared to setting or navigating, presumably linked to increased fish discards and waste released during hauling (Weimerskirch et al 2000; Abraham 2009). I expected that seabird abundance would increase after the commencement of hauling, once offal discarding began, attracting seabirds from the immediate area to the vessel, which would attract other seabirds from a wider area (Jiménez et al. 2011). There is a large increase in seabird abundance immediately after hauling commences, but numbers then remain more or less constant throughout the remainder of the haul. It is somewhat surprising that black-browed albatross numbers increased as the haul progressed as none of the other species numbers changed significantly as the haul progressed and species richness was also unaffected.

3.4.3 Bycatch susceptibility of the seabird assemblage

The south coast of South Africa experiences high levels of seabird bycatch from pelagic longline vessels while bycatch levels are significantly lower in international waters surrounding South Africa (Chapter 2) and few if any birds are killed in Mozambican waters (Chacate and Mutombene 2015; Mutombene 2015). Many of the seabird species regularly recorded off the Agulhas Bank were considered highly susceptible to longline bycatch. The White-chinned petrel, which is the species killed most frequently on pelagic longlines off South Africa (Chapter 2), was the most abundant bird at vessels off the Agulhas Bank, while large numbers of other species considered highly susceptible to bycatch were also recorded here. Only three other species regarded as highly susceptible to pelagic longline bycatch off southern Africa were recorded off the east coast, representing only 6% of total seabird abundance at vessels in this region.

Seabird bycatch is much higher off the Agulhas Bank (0.133 birds per 1000 hooks) compared to the international waters off the east coast (0.061 birds per 1000 hooks). The shy albatross is the most commonly caught albatross species, although both yellow-nosed sp. and black-browed are also recorded as bycatch (Table 3.3). Wandering-type albatross were only recorded as bycatch off the east coast, despite their being observed in larger numbers off the Agulhas Bank (Table 3.3).

Table 3.3 Summary of seabird species considered highly or moderately susceptible to seabird bycatch off South Africa. Bycatch rate (birds per 1000 hooks) and species composition (%CO) is summarised per region, and is a percentage of those species/species groups considered highly or moderately susceptible to seabird bycatch.

Species	Agulhas Bank			East Coast		
	Bycatch rate & composition (%)	Observed (%CO)	Daily mean	Bycatch rate & composition (%)	Observed (%CO)	Daily mean
White-chinned petrel	0.094 (70%)	40%	114	0.041 (68%)	31%	12
Shy Albatross	0.022 (16%)	8%	22	0.008 (14%)	4%	2
Black-browed albatross	0.004 (3%)	23%	65	-	17%	6
Yellow-nosed albatross spp.	0.008 (6%)	8%	23	0.003 (5%)	17%	6
Cape gannet	0.003 (2%)	< 1%	< 1	-	-	-
Giant petrel sp.	0.001 (1%)	< 1%	3	0.003 (5%)	2%	< 1
Wandering/Tristan albatross	-	< 1%	2	0.006 (9%)	1%	< 1
Royal albatross sp.	< 0.001 (< 1%)	< 1%	< 1	-	-	-
Brown skua	< 0.001 (< 1%)	< 1%	1	-	3%	1
Grey petrel	-	-	-	-	3%	< 1
Cape petrel	< 0.001 (< 1%)	14%	40	-	11%	4
Albatross unidentified	0.002 (< 1%)	-	-	-	-	-

Due to the absence of national jurisdiction, regulations pertaining to seabird bycatch are less rigorous when fishing in international waters. While fishing under South African fishing permits in international waters, vessels are allowed to set lines during daylight hours provided they use two bird-scaring lines and appropriate line weighting (DAFF 2013). My data show that the comparatively lower seabird bycatch rate off the east coast (in international waters) is most likely related to the lower seabird abundance; white-chinned petrel and shy albatross (the two most commonly recorded bycatch species in the region) averaged 10 times more abundant off the Agulhas Bank compared to the east coast. The reduced numbers of white-chinned petrels in particular (a species known to facilitate the bycatch of other larger species in particular, Jiménez et al. 2012), may help reduce bycatch levels off the east coast. Although relatively low numbers of wandering-type albatrosses are recorded as bycatch off South Africa (Chapter 2) or in adjacent international waters, these levels should not be considered

negligible as the worldwide population is considerably lower than other commonly recorded bycatch species.

Despite black-browed albatrosses being the most abundant albatross species off the Agulhas Bank (three times more abundant than shy albatrosses), three times more shy albatrosses are killed on longlines operating off the Agulhas Bank. This mismatch between abundance and bycatch is probably explained by body size and the associated dominance hierarchy of scavenging birds behind fishing vessels. The larger size of shy albatrosses compared to black-browed and Indian yellow-nosed albatrosses presumably ensures greater access to discards and bait (Jiménez et al. 2011). Wanless (1998) observed seabird dominance hierarchies behind deep-water hake trawlers off South Africa and noted that shy albatross was the most dominant of the mollymawk species while white-chinned petrels were the least efficient scavengers as they were regularly displaced from fisheries discards by other, larger species. Despite low numbers (< 1 per day) of wandering-type albatrosses observed off the east coast, they represented c. 10% of the seabird bycatch in the region. This mismatch is again likely to be explained by the dominance hierarchy, with the large wandering-type albatrosses able to outcompete other albatross and petrel species. Large bodied giant petrels are also able to outcompete other species behind fishing vessels (Jiménez et al. 2011), and thus it is perhaps surprising that they are recorded at such low bycatch rates off South Africa. However, my data suggest that their low bycatch rate can be explained by their low abundance around pelagic longline vessels off southern Africa.

It is important to note that almost all seabird bycatch from pelagic longliners occurs during line setting (Brothers et al. 2010) while all my seabird counts were conducted during line hauling. Therefore it is possible that discrepancies in species assemblage/bycatch data are explained by temporal variation of the seabird assemblage (Weimerskirch et al. 2000; Jiménez et al. 2011). As longlines were set during the night-time only off the Agulhas Bank with seabird observations made during daylight hauling operations, mismatches in species abundance/bycatch data could be explained by differences in foraging activity of seabird species. However species assemblage/bycatch ratios off the Agulhas Bank were similar to those data from the east coast, when line setting and seabird counts were performed during daylight hours (Table 3.3). Aside from the temporal differences in seabird assemblage there are likely to be differences in the composition of the assemblage due to the different activity of the vessel; different species and numbers may associate with hauling rather than setting and vice versa, which may explain these discrepancies between observed and bycaught birds.

Despite the high species richness off the Mozambique coast, few species attending vessels here are susceptible to pelagic longline bycatch and were only recorded in very small numbers. The paucity of susceptible species in Mozambican waters is likely the reason for the lack of seabird bycatch by pelagic longliners in this region. The most abundant species off the Mozambique coast, flesh-footed shearwater, has not been confirmed as seabird bycatch off southern Africa, despite being regularly caught off Australia (Baker and Wise 2005). This species was not recorded associating with longline vessels off the Agulhas Bank or east coast in my study, and is generally not known to accumulate in large numbers around fishing vessels in these areas (B Rose pers. comm.). However it does congregate behind fishing vessels off southern Mozambique and the east coast of South Africa, where other, larger species such as white-chinned petrels are much less abundant (Hockey et al. 2005).

By investigating the seabird assemblage associated with longline vessels throughout the southern oceans, trends in the spatial pattern of seabird bycatch can be better explained. Some species are regularly recorded as longline bycatch in certain areas, however are absent from the bycatch list absent in other areas, despite regularly occurring in the region. Longline vessels operating in the Mediterranean Sea regularly record Scopoli's shearwater (*Calonectris diomedea*) as pelagic longline bycatch (Báez et al. 2014; Cortéset al. 2017), while longline vessels operating off New Zealand and Australia regularly record sooty shearwaters (*Ardenna grisea*) as bycatch (Uhlmann 2003). Despite both sooty and Cory's shearwater (*C. borealis*; until recently considered conspecific with Scopoli's shearwater) occurring in large numbers in South African waters, neither of these species have been confirmed as longline bycatch off South Africa (Chapter 2). My results indicate that neither species attends pelagic longline vessels in large numbers off southern Africa, which is likely an important factor explaining their absence from local bycatch, although differences in gear configuration, fishing methods and mitigation measure use are all likely to play an important role too. Also, as with the flesh-footed shearwater discrepancy described above, it seems likely that these species' foraging ecology differs in their breeding and non-breeding ranges.

My study demonstrates the large spatial variation in seabird composition, richness and abundance off southern Africa, and hence helps explain the large spatial variation in seabird bycatch in the subregion. An understanding of the seabird species and abundances associated with longline vessels helps explain the composition of seabird bycatch from the fishery. To fully explain mismatches in seabird bycatch and seabird abundance data from an area, an understanding of the workings of the fishery as well as composition of the seabird assemblage associated with the vessels is required. Seasonal

variation in seabird abundance was not investigated in my study, however it is known to strongly influence seabird distributions and assemblages associated with fishing vessels (Olmos 1997; Gandini and Pon 2007; Jiménez et al. 2011) and will certainly influence the structure of the seabird assemblage off southern Africa.

An understanding of the factors influencing seabird bycatch is important when implementing seabird conservation plans. Seabird assemblage data associated with fishing vessels are useful when considering spatial conservation strategies such as fisheries closures (Grantham et al. 2008). These data could further contribute to our understanding of the seabird composition on the fishing grounds and how it is affected both spatially and temporally. These data help to explain the high levels of bycatch of certain seabird species off South Africa, as well as the absence of bycatch of other species prone to seabird bycatch in longline fisheries worldwide, which are useful when implementing species-level conservation plans.

Appendix 3.1. Species composition and abundance of daily counts (*n* = 41) of seabirds recorded off the Agulhas Bank with n representing total number and %FO the frequency of occurrence of seabirds. IUCN status was used as follows; Critically Endangered (CE), Endangered (EN), Vulnerable (VU), Near-Threatened (NT) and Least Concern (LC) as obtained from www.iucnredlist.org.

Species name	**Scientific name**	**IUCN status**	***n***	**Daily mean**	**Daily max**	**%FO**
Wandering/Tristan albatross	*Diomedea exulans/dabbenena*	VU/CE	76	2	21	61
Northern royal albatross	*Diomedea sandfordi*	EN	21	< 1	2	49
Southern royal albatross	*Diomedea epomophora*	VU	15	< 1	2	34
Shy-type albatross	*Thalassarche cauta/steadi*	NT	904	22	50	98
Salvin's albatross	*Thalassarche salvini*	VU	1	< 1	1	1
Black-browed albatross	*Thalassarche melanophris*	EN	2664	65	250	98
Atlantic yellow-nosed albatross	*Thalassarche chlororhynchos*	EN	27	< 1	10	17
Indian yellow-nosed albatross	*Thalassarche carteri*	EN	939	23	100	88
Yellow-nosed albatross spp.	*Thalassarche carteri/chlororhynchos*	EN	77	2	55	10
Giant petrel sp.	*Macronectes halli/giganteus*	LC	105	3	11	88
White-chinned petrel	*Procellaria aequinoctialis*	VU	4667	114	300	100
Spectacled petrel	*Procellaria conspicillata*	VU	2	< 1	1	5
Cape petrel	*Daption capense*	LC	1655	40	200	100
Great-winged petrel	*Pterodroma macroptera*	LC	2	< 1	1	5
Soft-plumaged petrel	*Pterodroma mollis*	LC	4	< 1	1	10
Blue petrel	*Halobaena caerulea*	LC	8	< 1	4	12
Sooty shearwater	*Ardenna gravis*	NT	2	< 1	1	5
Prion sp.	*Pachyptila* sp.	LC	13	< 1	5	15
Wilson's storm-petrel	*Oceanites oceanicus*	LC	4	< 1	2	7
Black-bellied storm-petrel	*Fregetta tropica*	LC	301	7	50	41
Brown skua	*Stercorarius antarctica*	LC	45	1	6	56
Cape gannet	*Morus capensis*	VU	6	< 1	2	12

Appendix 3.2. Species composition and abundance of daily counts (Vessel A = 16 days, Vessel B = 26 days) of seabirds recorded off the east coast of southern Africa with n representing total number and %FO the frequency of occurrence. IUCN status was used as follows; Critically Endangered (CE), Endangered (EN), Vulnerable (VU), Near-Threatened (NT) and Least Concern (LC) as obtained from www.iucnredlist.org.

Species name	**Scientific name**	**IUCN status**	***n***	**Daily mean**	**Daily max**	**%FO**
Vessel A						
Wandering/Tristan albatross	*Diomedea exulans/dabbenena*	VU/CE	16	1	8	25
Shy-type albatross	*Thalassarche cauta/steadi*	NT	58	4	40	44
Black-browed albatross	*Thalassarche melanophris*	EN	254	16	40	88
Grey-headed albatross	*Thalassarche chrysostoma*	EN	1	< 1	1	6
Indian yellow-nosed albatross	*Thalassarche carteri*	EN	242	15	45	100
Sooty albatross	*Phoebetria fusca*	EN	3	< 1	1	13
Light-mantled albatross	*Phoebetria palpebrata*	NT	1	< 1	1	6
Giant petrel sp.	*Macronectes halli/giganteus*	LC	8	< 1	3	38
White-chinned petrel	*Procellaria aequinoctialis*	VU	465	29	70	100
Grey petrel	*Procellaria cinerea*	NT	18	1	10	31
Cape petrel	*Daption capense*	LC	163	10	40	94
Great-winged petrel	*Pterodroma macroptera*	LC	4	< 1	1	25
Soft-plumaged petrel	*Pterodroma mollis*	LC	9	< 1	5	25
Sooty shearwater	*Ardenna gravis*	NT	4	< 1	4	6
Prion sp.	*Pachyptila* sp.	LC	135	8	50	75
Brown skua	*Stercorarius antarctica*	LC	33	5	8	56
Vessel B						
Wandering/Tristan albatross	*Diomedea exulans/dabbenena*	VU/CE	3	< 1	1	12
Shy-type albatross	*Thalassarche cauta/steadi*	NT	11	< 1	5	19
Black-browed albatross	*Thalassarche melanophris*	EN	14	< 1	5	23
Indian yellow-nosed albatross	*Thalassarche carteri*	EN	178	7	25	85
Light-mantled albatross	*Phoebetria palpebrata*	NT	1	< 1	1	4
Giant petrel sp.	*Macronectes halli/giganteus*	LC	30	1	5	54
White-chinned petrel	*Procellaria aequinoctialis*	VU	173	7	30	62
Grey petrel	*Procellaria cinerea*	NT	2	< 1	2	8
Cape petrel	*Daption capense*	LC	31	1	20	19
Great-winged petrel	*Pterodroma macroptera*	LC	21	1	5	38
Soft-plumaged petrel	*Pterodroma mollis*	LC	38	1	6	65
Sooty shearwater	*Ardenna gravis*	NT	1	< 1	1	4
Prion sp.	*Pachyptila* sp.	LC	131	5	50	62
Wilson's storm-petrel	*Oceanites oceanicus*	LC	1	< 1	1	6
South polar skua	*Stercorarius maccormicki*	LC	4	< 1	2	8
Brown skua	*Stercorarius antarctica*	LC	127	5	22	69

Appendix 3.3. Species composition and abundance of daily counts (*n* = 76) of seabirds recorded off the Mozambique coast of southern Africa with n representing total number and %FO the frequency of occurrence. IUCN status was used as follows; Critically Endangered (CE), Endangered (EN), Vulnerable (VU), Near-Threatened (NT) and Least Concern (LC) as obtained from www.birdlife.org.

Species name	**Scientific name**	**IUCN status**	***n***	**Daily mean**	**Daily Max**	**%FO**
Shy-type albatross	*Thalassarche cauta/steadi*	NT	1	< 1	1	1
Black-browed albatross	*Thalassarche melanophris*	EN	3	< 1	1	4
Indian yellow-nosed albatross	*Thalassarche carteri*	EN	96	1	40	16
Giant petrel sp.	*Macronectes halli/giganteus*	LC	4	< 1	2	4
White-chinned petrel	*Procellaria aequinoctialis*	VU	94	1	35	12
Cape petrel	*Daption capense*	LC	2	< 1	2	1
Great-winged petrel	*Pterodroma macroptera*	LC	25	< 1	12	11
Soft-plumaged petrel	*Pterodroma mollis*	LC	14	< 1	5	4
Flesh-footed shearwater	*Ardenna carneipes*	LC	458	6	300	20
Sooty shearwater	*Ardenna gravis*	NT	2	< 1	1	3
Prion sp.	*Pachyptila* sp.	LC	202	3	100	8
Wilson's storm-petrel	*Oceanites oceanicus*	LC	154	2	50	37
Black-bellied storm-petrel	*Fregetta tropica*	LC	250	3	50	38
Swinhoe's storm-petrel	*Oceanodroma monorhis*	NT	2	< 1	1	3
Matsudaira's storm-petrel	*Oceanodroma matsudairae*	VU	2	< 1	1	3
White-faced storm-petrel	*Pelagodroma marina*	LC	1	< 1	1	1
Red-footed booby	*Sula sula*	LC	28	< 1	2	32
Great frigatebird	*Fregata minor*	LC	246	3	50	74
South polar skua	*Stercorarius maccormicki*	LC	7	< 1	3	5
Brown skua	*Stercorarius antarctica*	LC	32	< 1	10	7
Kelp gull	*Larus dominicanus*	LC	1	< 1	1	1
Arctic tern	*Sterna paradisaea*	LC	29	< 1	12	4

CHAPTER 4

YEAR-ROUND MOVEMENTS OF WHITE-CHINNED PETRELS FROM MARION ISLAND, SOUTHWESTERN INDIAN OCEAN

Abstract

White-chinned petrels are proficient divers which compete aggressively for bait and offal at fishing vessels; as such they are the most frequently recorded procellariiform species in Southern Hemisphere longline fisheries bycatch. Previous genetic and movement studies have found very little overlap of foraging areas between *P. a. steadi* from the New Zealand sub-Antarctic islands and nominate *P. a. aequinoctialis* from South Georgia and sub-Antarctic islands in the southwestern Indian Ocean. My study investigated the year-round movements of 10 adult white-chinned petrels from Marion Island tracked with GLS loggers for 3 years. In addition, 20 GPS tracks were obtained from breeding white-chinned petrels during incubation (*n* = 9) and chick-rearing (*n* = 11). Of the GLS-tagged birds, six males and four females were initially tagged as non-breeders, but most bred in subsequent years. All GLS-tagged birds remained in the area between southern Africa and Antarctica and did not make any major east/west movements; there was little variation in individual foraging ranges in successive years. Three core areas (50% kernels) were utilised; around PEI, c. 1000 km west of PEI and around South Africa, with some birds reaching southern Angola (15^{o}S) and south to 63^{o}S in Antarctic waters. Both sexes utilised similar foraging areas, with no difference in home range size (95% kernels). During incubation birds centred their foraging activity around PEI, chick-rearing birds ranged wider with core areas from PEI to South Africa, while non-breeding birds occupied two core areas, around PEI and South Africa. GPS-tagged birds foraged in similar areas to the GLS-tagged birds, ranging more widely during chick-rearing than during incubation. However, both trip duration and distance covered were slightly longer during incubation than chick-rearing. High levels of white-chinned petrel activity overlapped with intense longline fishing effort off the Agulhas Bank, with little overlap elsewhere. My results confirm the lack of foraging overlap between the two subspecies, and that nominate birds from South Georgia and the southwestern Indian Ocean forage in separate areas and thus should be treated as separate management units. Knowledge of the year-round movements of a vagile species such as the white-chinned petrel is important for its continued conservation.

4.1 Introduction

The white-chinned petrel is a large burrow-nesting petrel breeding on a number of sub-Antarctic islands with a worldwide population of some 1.2 million breeding pairs (c. 3 million mature birds; IUCN 2016). It comprises two subspecies; *P. a. steadi* breeds at the New Zealand sub-Antarctic islands and the nominate subspecies breeds on South Georgia (southwest Atlantic Ocean) and sub-

Antarctic islands in the southwestern Indian Ocean (Techow et al. 2009). The nominate subspecies comprises some 681 000 breeding pairs on South Georgia (Martin et al. 2009), 36 000 on PEI (Ryan et al. 2012), 23 600 on the Crozet Islands (Barbraud et al. 2008), 234 000 on the Kerguelen Islands (Barbraud et al. 2009), and < 100 pairs on the Falkland Islands (Reid et al. 2007). Population estimates of *P. a. steadi* are less accurate, but are thought to number 153 000 pairs on Disappointment Island (Rexer-Huber et al. 2016), 10 000 on Campbell Island (ACAP 2012) and 58 000 on the Antipodes (ACAP 2012). Populations on a number of breeding islands are thought to be decreasing, including those on South Georgia and Ile de la Possession in the Crozets (Berrow et al. 2000a; Barbraud et al. 2008). Due to these population reductions, attributed mainly to the impact of longline fishing, the white-chinned petrel is listed as Vulnerable by the IUCN (IUCN 2016).

White-chinned petrels are the most commonly caught seabird species in the southern oceans on both pelagic (DeLord et al. 2005; Petersen et al. 2009a; Ryan et al. 2012) and demersal longlines (Barnes et al. 1997; Nel et al. 2002; Barbraud et al. 2009). Their deep-diving capabilities (Chapter 5) relative to albatrosses makes them more susceptible to longline bycatch, as they are able to retrieve baited hooks from greater depths than shallow-diving species (Jiménez et al. 2012). They also have been implicated in the bycatch of other seabird species, as they retrieve baited hooks from beyond the diving capabilities of larger species, only to be displaced from the bait on returning to the surface (Jiménez et al. 2012). They frequently scavenge behind fishing vessels for discards and offal which can form a significant dietary component (Catard et al. 2000). Phillips et al. (2006) found a large overlap between the distribution of white-chinned petrels from South Georgia and fishing fleets operating off South America.

White-chinned petrels undertake vast foraging trips during both the breeding and non-breeding seasons, ranging from the tropics to the ice edge (Weimerskirch et al. 1999). Birds have been tracked with both GLS loggers and GPS loggers from a number of their breeding islands. Year-round movements of white-chinned petrels have been studied for birds breeding at South Georgia and Kerguelen Island using GLS loggers. All South Georgian birds migrated to the Patagonian Shelf and shelf-break waters off South America (Phillips et al. 2006), whereas Kerguelen birds headed south to Antarctic waters or northwest to the Benguela Current off South Africa (Delord et al. 2010a; Péron et al. 2010). White-chinned petrels breeding at the Crozet Islands visited the South African continental shelf but also ventured south to 65^{o}S in Antarctic waters, a maximum distance from the nest of 2300 and 3500 km, respectively (Weimerskirch et al. 1999). All these tracking studies are of the nominate subspecies; the only study of movements by *P. a. steadi* revealed that birds from New Zealand's

Antipodes Islands disperse to the west coast of South America during the non-breeding season (Somner et al. 2010).

The spatial segregation of different breeding colonies is not restricted to adult birds. By analysing DNA markers of white-chinned petrels killed as fisheries bycatch in South African and New Zealand fisheries, Techow et al. (2016) concluded that all birds killed in the South African fishery were of the nominate subspecies while all those killed by the New Zealand fishery were *P. a. steadi*. This evidence along with results of tracking studies suggests that all age groups of the two subspecies forage in different regions, with the only potential overlap occurring between New Zealand and South Georgian birds whilst foraging off the west coast of South America (Phillips et al. 2006; Somner et al. 2010).

My study reports the year-round movements of white-chinned petrels from PEI. Adults equipped with GLS loggers were tracked for three years, which allowed us to investigate variation in foraging zones between individuals, seasons and years, as well as determine the overlap between key foraging areas and longline fishing effort. Given their preponderance in fisheries bycatch as well as their deep-diving abilities often resulting in the secondary hooking of larger species (Jiménez et al 2012), an understanding of the year-round movements of white-chinned petrels has important conservation implications for the management of regional fisheries.

4.2 Methods

GLS loggers were deployed on 20 white-chinned petrels caught in a colony close to the research station on sub-Antarctic Marion Island (46°52'S, 37°51'E) towards the end of the incubation period from 18–23 December 2009. The GLS devices (MK7, British Antarctic Survey, Cambridge) were attached to plastic leg bands and weighed 4 g (~0.3% body mass), well below the recommended 3% limit for flying birds (Phillips et al. 2003). Birds were sexed from culmen length and depth (Ryan 1999). The breeding status of birds was inferred by examining GLS light data; continuous periods of darkness lasting 2–3 weeks during incubation were assumed to be birds incubating eggs in their burrows. Unfortunately I was unable to ascertain whether birds raised chicks successfully. When birds were caught in 2009 they were on the surface at night, not in breeding burrows, and subsequent examination of light traces indicated that only one was breeding (the others were either non-breeders or had failed prior to deployment).

To retrieve the devices, tagged birds were searched for within the same colony in 2010–2012, but the only birds recaptured were recovered during the pre-laying period in 2012 (02–20 October),

when a determined effort was made to recover devices. The different breeding stages were defined as pre-breeding (October), incubation (November–December), early chick-rearing (January), late chick-rearing (February–April) and non-breeding (May–September).

GLS loggers are much less accurate (~200 km accuracy) than GPS loggers or other tracking devices and only provide two location estimates per day, but they can provide long-term data on seabird movements due to their long battery life (Phillips et al. 2004). They record maximum ambient light levels every ten minutes, which are used to determine local sunrise and sunset and thus estimate location every 12 h (Wilson et al. 1992). The loggers were calibrated for a minimum of 20 days before and after deployment. The loggers tested for saltwater immersion every 3 sec with temperature only recorded if the logger was continuously submerged for 20 min.

To augment the GLS tracks, GPS devices (CatTraqTM; 16 Mb memory; 230 mA lithium-ion battery; Mr Lee Technologies) measuring 45.7 x 30.5 x 12.7 mm and weighing 25 g were deployed on white-chinned petrels breeding on Marion Island during the late incubation (*n* = 11; December 2012) and chick-rearing (*n* = 13; January/February 2013) periods (roughly 2 months after GLS devices were retrieved). The GPS loggers were programmed using @trip PC (Version 2.0) to sample a position every 60 min, which allowed the batteries to last several weeks. Some of the birds also carried small TDR devices (2.7 g; 35.5 x 9 x 11.5 mm) in addition to the GPS devices; one bird during incubation and six during chick-rearing (see Chapter 5).

I used two software packages to analyse GLS data; BASTrack (British Antarctic Survey) was used to download and decompress the raw data, and a modified version of the R package *TripEstimation* (R Core Team 2015) written specifically to analyse GLS data from large Procellariiformes (J Cleeland pers. comm.) was used to analyse the GLS tracks. The sun elevation angle (when light levels change rapidly during twilight events) was obtained from the calibration period and used for the remainder of the analysis. All apparent outliers in the unfiltered locations were examined individually. Outliers might be generated by shading of the GLS when the bird was on water or when legs were tucked in under feathers during flight. During equinox periods latitudinal uncertainty is compromised, as day length depends weakly on latitude at this time (Ekstrom 2004). Therefore due to suspect latitude estimates during equinox periods, data 10 days either side of the equinoxes were excluded from analyses. *TripEstimation* compensates for shading of GLS devices and uses Markov Chain Monte Carlo simulation methods in a Bayesian framework to produce the most likely path from corrected positions with variables such as sea temperature and speed.

I examined predicted locations using the *AdehabitatHR* package (Calenge 2006) to create kernel Utilization Distribution estimates. A smoothing parameter (*h*) was chosen based on the reference bandwidth method for unsmoothed GLS data, with grid size set as 1 km. I encompassed 95% and 50% of locations to estimate home range and core range of the tracked white-chinned petrels, respectively (Wood et al. 2000). Recorded GPS tracks were analysed to obtain detailed information on the foraging movements of the petrels. Home range maps were then created using the geographic information system ArcGIS version 10.2 (ESRI, Redlands, USA). A trip was defined as the period between leaving and returning to the colony with only completed trips used for the analysis of trip duration (number of days) and distance (cumulative distance between positional fixes). Values given are means ± standard deviation, unless otherwise stated.

Pelagic longline fishing effort from the Indian and South Atlantic Oceans were obtained from IOTC (www.iotc.org/data/datasets) and ICCAT (www.iccat.int/en/accesingdb.HTML). Demersal (toothfish) longline data for areas controlled by the CCAMLR (Indian Ocean; south of 45^{o} S, Atlantic Ocean; south of 50^{o} S) were provided by CCAMLR. Longline fishing effort from December 2009 to October 2012 was summarised into 5^{o} x 5^{o} grid cells.

4.3 Results

Of the 20 GLS devices deployed, 10 were retrieved with useable data from six male and four female white-chinned petrels, after an average of 1022 ± 8 d (range 1013–1032 d; Table 4.1). Seven birds bred during the three-year study period, suggesting that at least some birds tracked were pre-breeders. Of the 24 GPS devices deployed, 20 devices (10 females, 10 males) were retrieved with useable data for an average of 22.9 ± 5.7 d (range 14.8–33.2 d; Table 4.1); nine during late incubation (December; 19.7 ± 5.9 d; range 14.8–33.0 d) and 11 during chick-rearing (eight early chick-rearing in January, and three late chick-rearing in February; average 25.6 ± 4.0 d (range 20.4–33.2 d).

Table 4.1. Summary of GLS (December 2009 – October 2012) and GPS (December 2012 – February 2013) deployments of white-chinned petrels from Marion Island.

Bird ID (sex)	Date device deployed	Date device retrieved	Period covered	Tracking duration (d)
Bird 1 (M)	18 Dec 2009	02 Oct 2012	Year-round	1019
Bird 2 (M)	18 Dec 2009	02 Oct 2012	Year-round	1019
Bird 3 (F)	18 Dec 2009	03 Oct 2012	Year-round	1020
Bird 4 (M)	18 Dec 2009	15 Oct 2012	Year-round	1032
Bird 5 (F)	23 Dec 2009	02 Oct 2012	Year-round	1014
Bird 6 (F)	23 Dec 2009	02 Oct 2012	Year-round	1014
Bird 7 (F)	23 Dec 2009	02 Oct 2012	Year-round	1014
Bird 8 (M)	23 Dec 2009	19 Oct 2012	Year-round	1031
Bird 9 (M)	23 Dec 2009	19 Oct 2012	Year-round	1031
Bird 10 (M)	23 Dec 2009	20 Oct 2012	Year-round	1032
GLS mean ± SD	-	-	-	*1023 ± 8*
Bird A (M)	05 Dec 2012	20 Dec 2012	Incubation	14.8
Bird B (F)	05 Dec 2012	20 Dec 2012	Incubation	14.8
Bird C (F)	05 Dec 2012	26 Dec 2012	Incubation	20.8
Bird D (M)	09 Dec 2012	24 Dec 2012	Incubation	15.0
Bird E (M)	09 Dec 2012	26 Dec 2012	Incubation	17.1
Bird F (F)	09 Dec 2012	26 Dec 2012	Incubation	17.1
Bird G (F)	09 Dec 2012	30 Dec 2012	Incubation	21.2
Bird H (M)	09 Dec 2012	01 Jan 2013	Incubation	23.2
Bird I(F)	09 Dec 2012	11 Jan 2013	Incubation	33.0
Bird J (F)	01 Jan 2013	21 Jan 2013	Early chick-rearing	20.4
Bird K (M)	01 Jan 2013	21 Jan 2013	Early chick-rearing	20.4
Bird L (M)	01 Jan 2013	27 Jan 2013	Early chick-rearing	25.4
Bird M (F)	01 Jan 2013	27 Jan 2013	Early chick-rearing	25.4
Bird N(F)	01 Jan 2013	27 Jan 2013	Early chick-rearing	25.4
Bird O (M)	01 Jan 2013	28 Jan 2013	Early chick-rearing	26.9
Bird P (F)	01 Jan 2013	08 Feb 2013	Early chick-rearing	31.1
Bird Q (F)	04 Jan 2013	06 Mar 2013	Early chick-rearing	26.6
Bird R (M)	04 Feb 2013	28 Feb 2013	Late chick-rearing	24.2
Bird S (M)	04 Feb 2013	10 Mar 2013	Late chick-rearing	33.2
Bird T (M)	06 Feb 2013	28 Feb 2013	Late chick-rearing	22.3
GPS mean ± SD	-	-	-	*22.9 ± 5.7*

4.3.1 Foraging locations

All birds remained in the western Indian, eastern South Atlantic or Southern Ocean between southern Africa and Antarctica. Three distinct core areas (50% use kernels) were identified for the ten GLS-tagged birds; around PEI, c. 1000 km west of PEI at approximately 47°S, 25°E and along the Agulhas Bank off South Africa (Figure 4.1). However, use of these core areas varied throughout the year (Figure 4.2). Two individuals made foraging trips up the west coast of southern Africa as far

north as southern Angola, while many of the birds foraged in areas southwest of Marion Island along the Southwest Indian Ocean Ridge in the region of the Southern Antarctic Circumpolar Current Front (Figure 4.1). Only one individual (Bird 4) foraged in Antarctic waters as far south as 63°S.

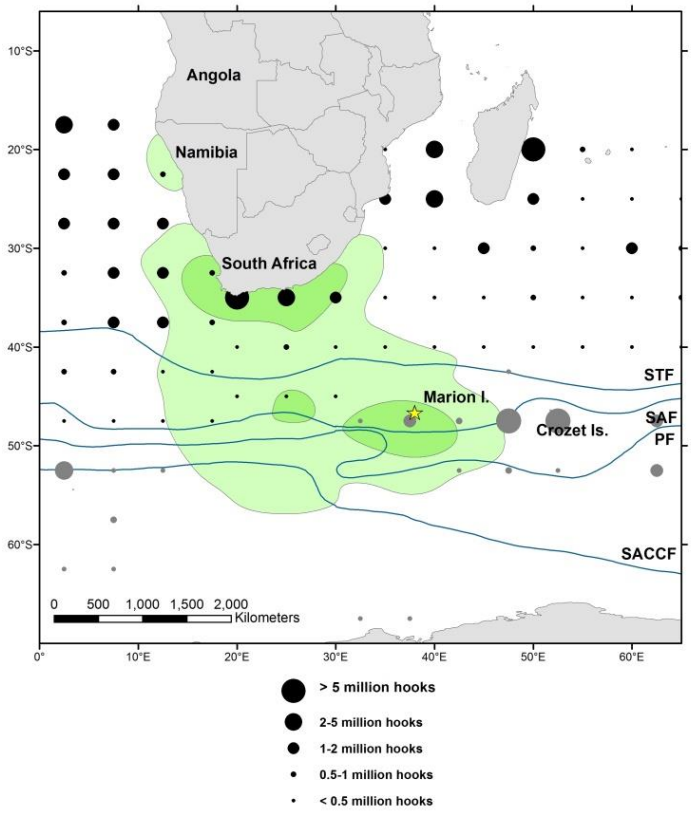


Fig. 4.1. 50% (dark green) and 95% (light green) density kernels of 10 adult white-chinned petrels from Marion Island, with total pelagic longline fishing effort (black circles) and demersal (toothfish) longline fishing effort (grey circles) per 5° x 5° square between December 2009 and October 2012. ☆ = Breeding colony on Marion Island, STF = Subtropical Front, SAF = Sub-Antarctic Front, PF = Polar Front, and SACCF = Southern Antarctic Circumpolar Current Front.

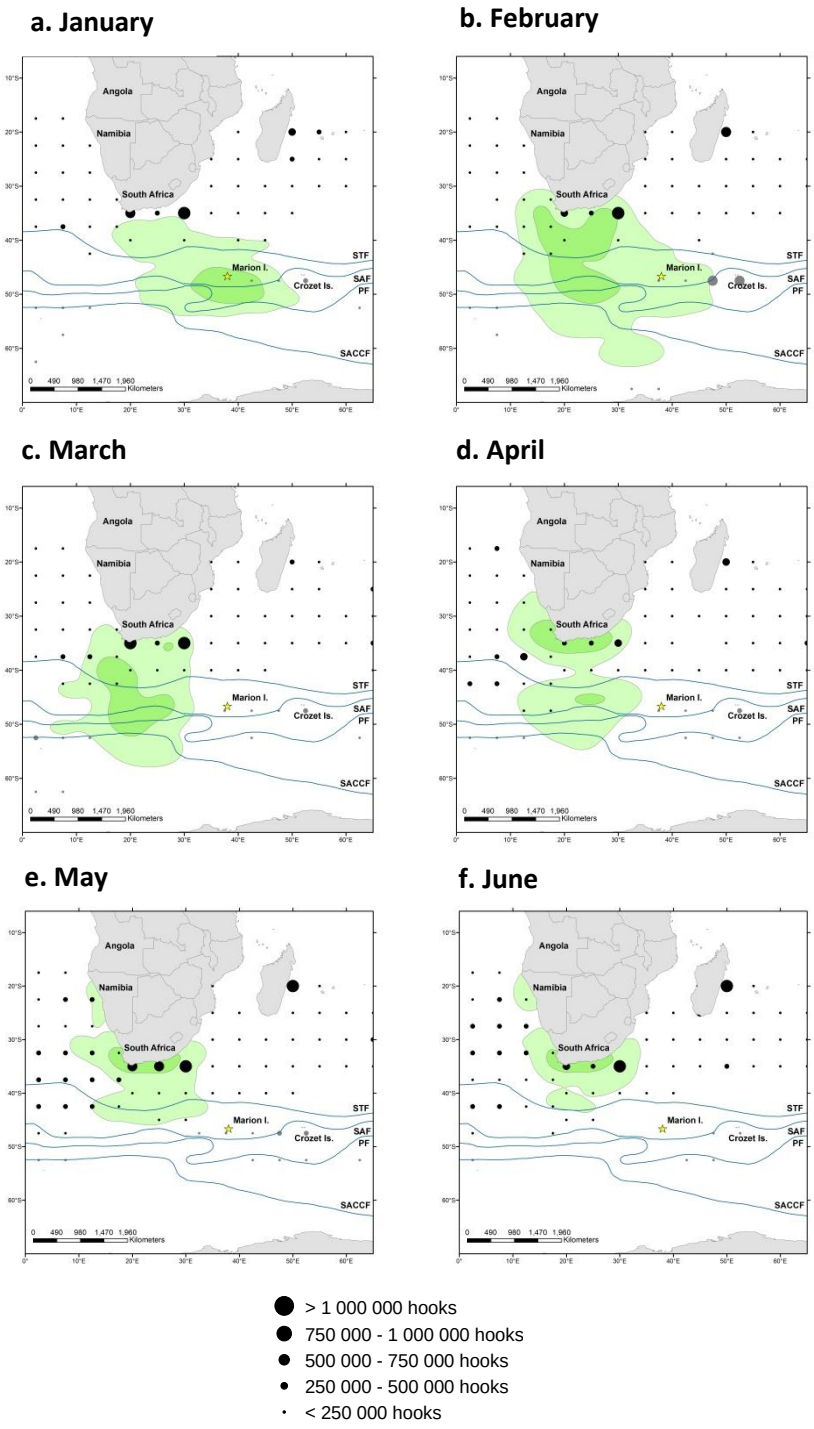


Fig. 4.2. (a–l) 50% (dark green) and 95% (light green) density kernels of adult white-chinned petrels from Marion Island per month, with total pelagic longline fishing effort (black circles) and demersal (toothfish) longline fishing effort (grey circles) per 5^{o} x 5^{o} square between December 2009 and October 2012. ☆ = Breeding colony on Marion Island, STF = Subtropical Front, SAF = Sub-Antarctic Front, PF = Polar Front, and SACCF = Southern Antarctic Circumpolar Current Front. Due to the

exclusion of data around equinox periods, kernels for March and September are based on considerably fewer location estimates.

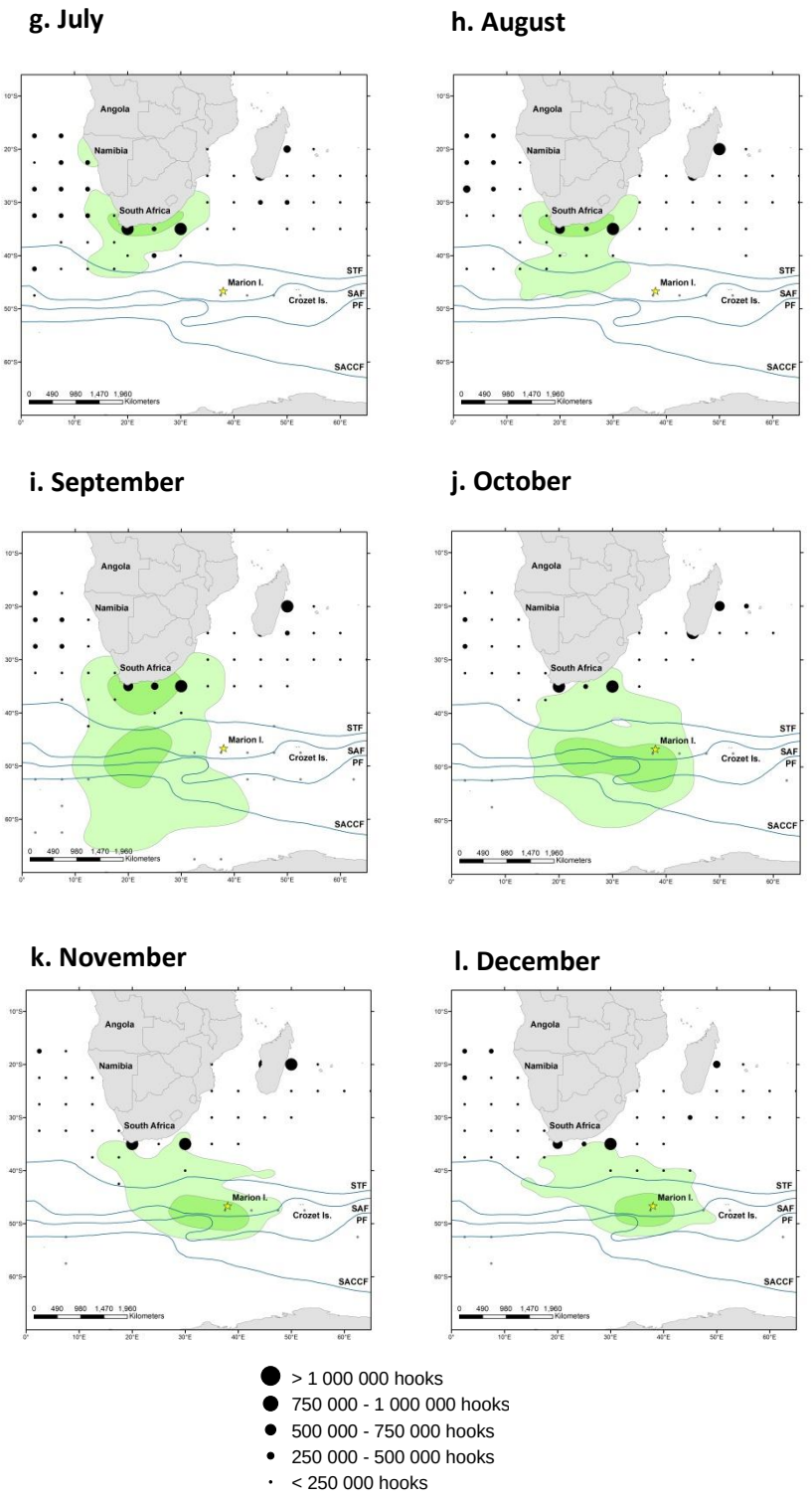


Fig. 4.2. (g-l) continued

Males and females visited similar areas (Appendix 4.2), with no significant difference in 95% foraging ranges (t_9 = 2.78; p = 0.232), despite males having slightly greater foraging ranges (5.4 million ± 0.9 million km^2) than females (4.1 million ± 1.6 million km^2). Both sexes utilized the same three core areas (50% kernels; Appendix 4.1), and had fairly high levels of overlap throughout the study (66%).

4.3.2 Inter-annual variation

Individual white-chinned petrels typically utilised similar foraging areas between years (Figure 4.3; Appendix 4.1). Apart from two individuals (Bird 6 and 10) which visited northern Namibia/southern Angola in 2010 and 2012 but not in 2011 (Appendix 4.2), most individuals visited similar areas and did not differ greatly between years. Despite combined home ranges averaging larger in 2010 (5.4 million km^2), than 2011 (4.3 million km^2) or 2012 (4.6 million km^2), the differences among years were not significant (ANOVA; F = 1.48; df = 2; p = 0.247; Table 4.2).

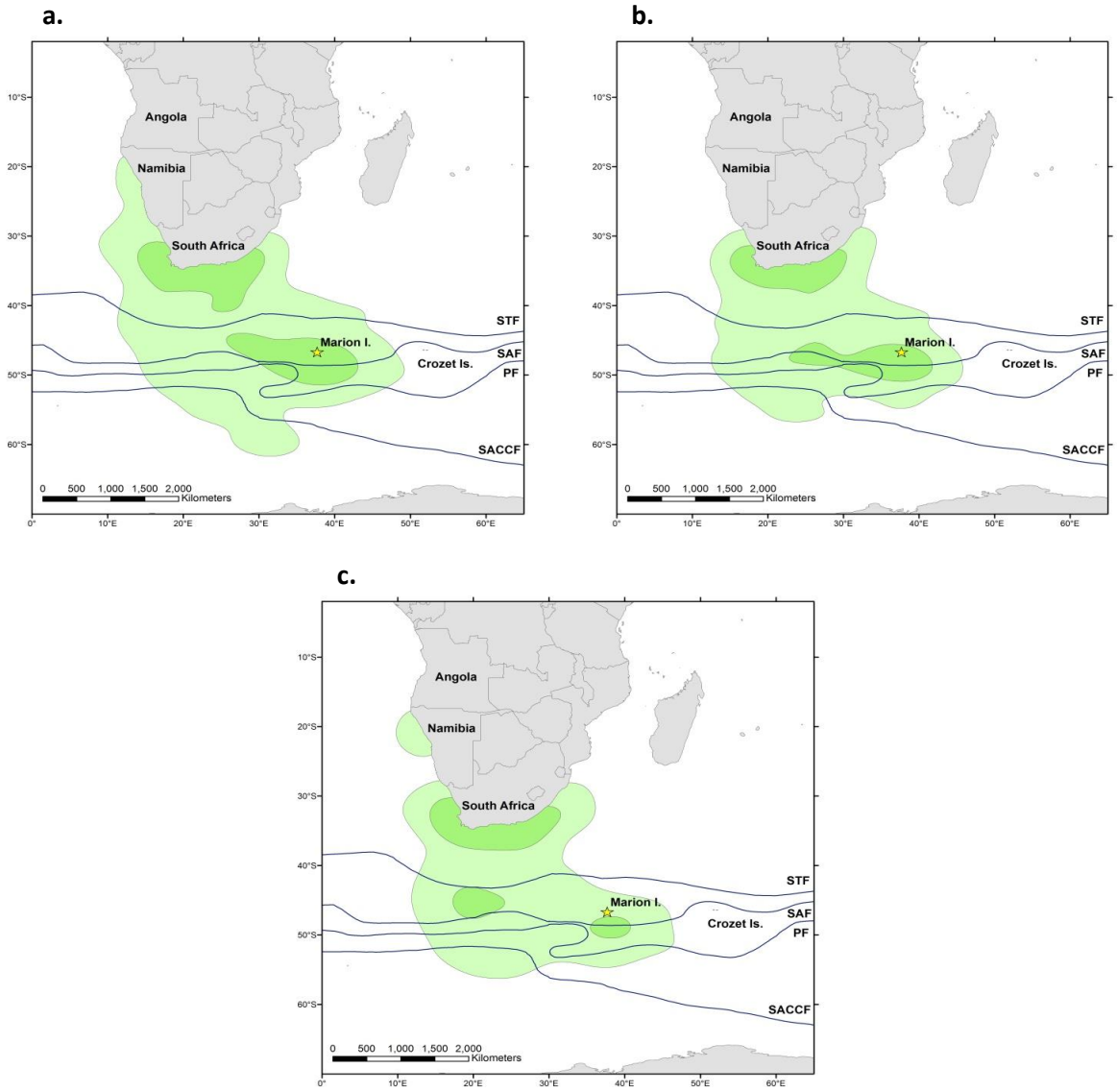


Fig. 4.3. 50% (dark green) and 95% (light green) density kernels of white-chinned petrels from Marion Island, from a. 2009/2010, b. 2011 and c. 2012. ☆= Breeding colony on Marion Island, STF = Subtropical Front, SAF = Sub-Antarctic Front, PF = Polar Front, and SACCF = Southern Antarctic Circumpolar Current Front.

Table 4.2. Summary of yearly area and overlap of 95% contours of white-chinned petrels from Marion Island.

Bird ID	2010 area (million km^2)	2011 area (million km^2)	2012 area (million km^2)	All years (million km^2)	Overlap (%); 2010–2011	Overlap (%); 2011–2012	Overlap (%); 20102012
1	4.2	5.2	4.2	4.4	78	77	76
2	5.9	4.9	4.3	5.4	82	73	67
3	2.9	3.1	2.3	2.8	86	62	65
4	6.2	4.1	6.4	5.7	66	62	62
5	6.1	3.6	7.4	6.1	52	43	70
6	6.6	3.6	3.5	4.8	48	65	53
7	2.7	3.0	3.4	3.0	67	64	70
8	6.3	5.7	4.5	5.8	65	55	52
9	4.9	4.3	5.1	4.5	82	74	74
10	8.0	5.9	4.5	6.8	64	72	49
Mean ± SD	5.3 ±1.4	4.3 ± 1.2	4.6 ± 1.2	4.9 ± 1.3	67	64	63
All birds	6.7	5.2	5.7	6.6	63	82	74

4.3.3 Effect of breeding and non-breeding stages

Movement data were retrieved from two complete breeding seasons (October–April) and one near-complete breeding season (December–April) between December 2009 and October 2012. Of the ten GLS-tagged birds from which data were successfully retrieved only one bird bred in all three breeding seasons; three birds did not breed at all, with the remaining six birds breeding once or twice throughout the study period (Table 4.3).

Table 4.3. Summary of breeding status of the ten white-chinned petrels tracked with GLS loggers.

Bird ID (sex)	2009/10 breeding season	2010/11 breeding season	2011/12 breeding season
1 (M)	Non-breeding	Breeding	Breeding
2 (M)	Non-breeding	Non-breeding	Non-breeding
3 (F)	Non-breeding	Non-breeding	Breeding
4 (M)	Non-breeding	Non-breeding	Non-breeding
5 (F)	Non-breeding	Breeding	Breeding
6 (F)	Breeding	Breeding	Breeding
7 (F)	Non-breeding	Non-breeding	Breeding
8 (M)	Non-breeding	Breeding	Breeding
9 (M)	Non-breeding	Non-breeding	Non-breeding
10 (M)	Non-breeding	Non-breeding	Breeding

Breeding birds (combined pre-breeding, incubation and chick-rearing stages) ranged widely, with core areas (50% contours) concentrated from PEI up to 1500 km west, as well as along the south and southwest coast of South Africa. During the pre-breeding stage (October) foraging activity was

centred over PEI, however these birds also ranged widely, as far as the southeast coast of South Africa (Figure 4.4a). Most incubating birds' (November–December) foraging activity was centred over PEI (Figure 4.4b) with only one bird foraging off the southeast coast of South Africa. Early chick-rearing birds (January) occupied a similar range to incubating birds, and were centred around PEI (Figure 4.4c), while late chick-rearing birds (February–April) occupied a much larger range with core areas along the South African continental shelf and approximately 1500 km west of PEI (south of the Sub-Antarctic Front, Figure 4.4d). Non-breeding birds (May–September; Figure 4.22e–i) ranged even further afield with most activity concentrated in South African waters; two birds dispersed as far as north as southern Angola and one bird reached southern Mozambique. Late in the non-breeding season (September), foraging activity was much further dispersed with birds ranging from the South African coast to Antarctic waters (Figure 4.2i).

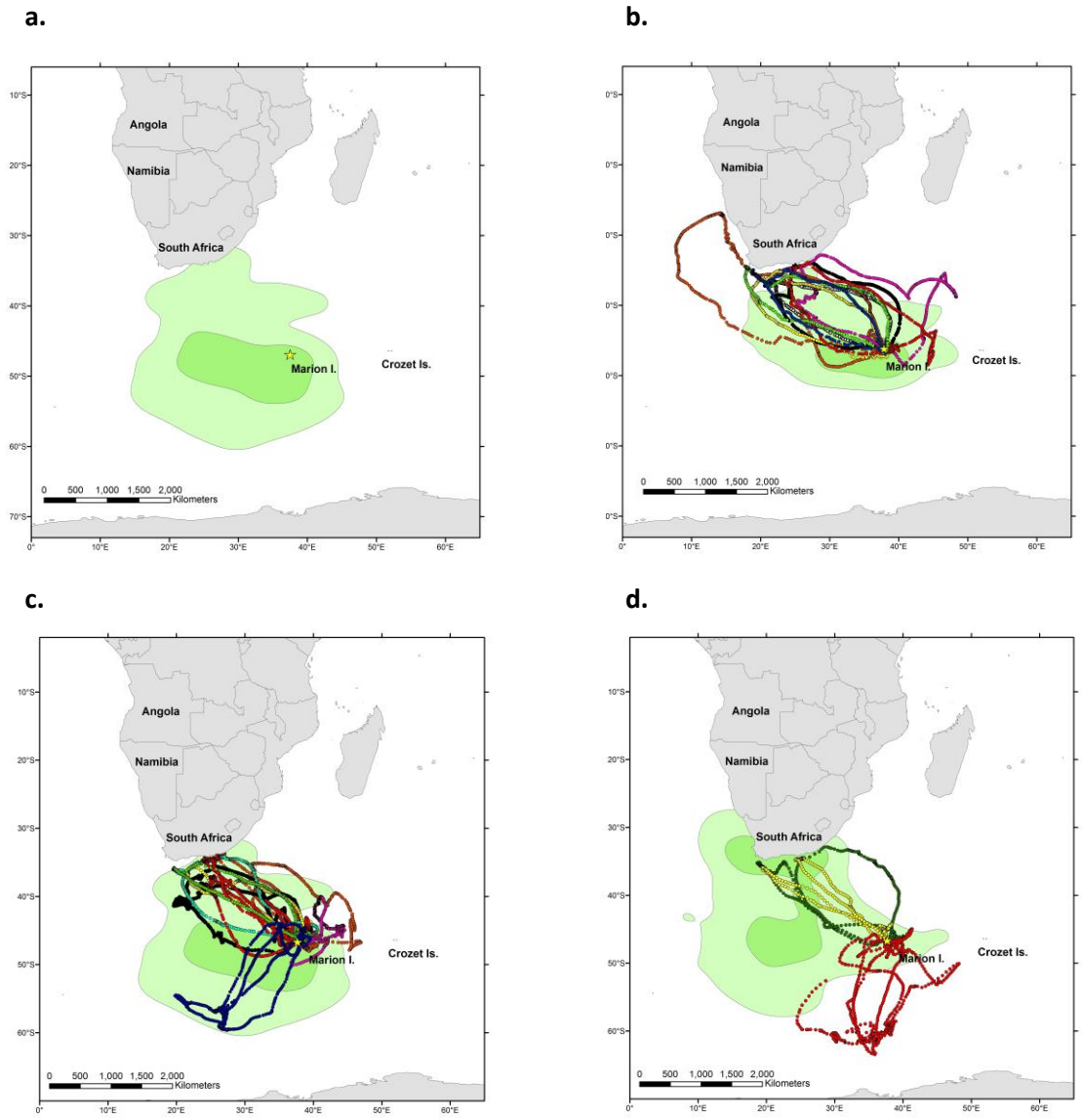


Fig. 4.4. 50% (dark green) and 95% (light green) density kernels and GPS tracks (coloured circles) of white-chinned petrels from Marion Island during a. pre-breeding (October), b. incubation (November–December; GPS birds = 9), c. early chick-rearing (January; GPS birds = 8), and d. late chick-rearing periods (February–April; GPS birds = 3). ☆ = Breeding colony on Marion Island, STF = Subtropical Front, SAF = Sub-Antarctic Front, PF = Polar Front, and SACCF = Southern Antarctic Circumpolar Current Front.

Of the GPS-tracked birds, incubating birds averaged longer trips (9.1 ± 6.4 d) than chick rearing birds (6.1 ± 4.4 d; Table 4.4) although this difference was insignificant (*T* test; T_{49} = 1.193; p = 0.061). Incubating birds also covered greater distance per trip (4311 ± 3104 km, maximum 9181 km) than chick-rearing birds (3197 ± 2452 km; maximum 7007 km; Table 4.4) however again this difference

was insignificant (*T* test; T_{49} = 1.349; p = 0.184). While undertaking foraging trips, distance covered per day was similar between incubating (470 ± 82 km per day) and chick-rearing birds (501 ± 99 km per day; *T* test; T_{20} = 0.746; p = 0.465). During incubation all GPS-tracked birds visited South African waters with one bird ranging up the west coast as far as southern Namibia (Figure 4.4b). Chick-rearing birds either visited the south coast of South Africa or made trips southwest of Marion Island into Antarctic waters as far south as 63° S (Figure 4.4c and 4.4d), with no individuals visiting both South African and Antarctic waters. Of the short trips (< 3 d duration), 65% (*n* = 17) were to an area c. 200 km north or northeast of Marion Island.

Table 4.4. Summary of white-chinned petrel foraging trips during incubation and chick rearing.

Bird	No. of trips	Trip duration (d)			Trip distance (km)		
		Mean ± SD	Max	Min	Mean ± SD	Max	Min
Incubating	*16*	*9.1 ± 6.4*	*18.8*	*0.8*	*4311 ± 3104*	*9181*	*218*
A	1	14.4	–	–	7224	–	–
B	1	14.4	–	–	5448	–	–
C	4	4.5 ± 4.4	12.7	0.8	2185 ± 2501	5551	218
D	1	14.6	–	–	5757	–	–
E	1	12.3	–	–	5579	–	–
F	1	14.4	–	–	5827	–	–
G	1	18.8	–	–	9181	–	–
H	1	11.8	–	–	5909	–	–
I*	5	4.4 ± 5.5	11.8	0.8	2750 ± 3927	8610	252
Chick-rearing	*37*	*6.1 ± 4.4*	*13.0*	*0.8*	*3197 ± 2452*	*7007*	*220*
J	3	4.7 ± 2.3	7.2	2.8	2054 ± 1135	3363	1341
K	3	4.5 ± 6.1	11.6	0.9	2080 ± 3153	5721	220
L	3	6.3 ± 5.3	12.2	2.0	3847 ± 3130	7007	747
M	3	4.8 ± 5.9	11.6	0.9	2262 ± 2971	5691	479
N	1	11.8	–	–	6158	–	–
O	3	7.6 ± 2.4	9.7	5.0	4290 ± 685	4831	3520
P*	3	8.4 ± 4.9	13.0	3.8	3741 ± 4352	6818	664
Q*	5	6.1 ± 4.4	10.8	1.7	3383 ± 2741	5822	824
R	4	6.0 ± 5.4	11.3	1.0	2871 ± 2775	5520	292
S*	7	4.0 ± 4.0	9.0	0.8	2616 ± 2695	6530	519
T	2	11.2 ± 0.3	11.4	11.0	5377 ± 535	5755	4998
All birds	*53*	*7.0 ± 5.1*	*18.8*	*0.8*	*3538* ± 2686	*9181*	*218*

* Represents birds which undertook a trip which was not fully covered by the GPS logger.

4.3.4 Overlap with fisheries

Longline fishing effort during the study period was greatest in the IOTC area (168.6 million hooks set; 59% of the total effort), with a maximum mean monthly effort per 5° x 5° grid cell of 0.8 million

hooks set in May. Fishing effort in the ICCAT area was barely half that in the IOTC (87.1 million hooks; 30% of total effort) and even less in the CCAMLR area (31.5 million hooks; 11% of total effort). Maximum mean monthly fishing effort per 5^{o} x 5^{o} square in these areas occurred in April (0.5 million hooks) and February (1.4 million hooks), respectively. Overall there were low levels of overlap between white-chinned petrel foraging areas and fishing effort (Figure 4.1). The only overlap between areas of high utilisation by white-chinned petrels (50% use kernel distributions) and intense longline fishing effort (> 0.5 million hooks per month) occurred off the Agulhas Bank during the non-breeding season (May–September; Figure 4.2).

4.4 Discussion

My study is the first to report the year-round movements of white-chinned petrels from PEI. None of the tracked adults made any significant east/west movements; all remained in the southwestern Indian, southeastern Atlantic or adjacent Southern Ocean between southern Africa and Antarctica. The two largest core areas utilised, around PEI and Agulhas Bank of South Africa, are highly productive waters (Kaehler et al. 2000; Shannon and O'Toole 2003) with abundant prey for white-chinned petrels. Several birds also visited eddies spawned over the Southwest Indian Ocean Ridge, southwest of Marion Island, which are key foraging areas for other large marine predators such as grey-headed albatrosses (*T. chrysostoma*) and southern elephant seals (*Mirounga leonina*; Jonker and Bester 1998; Nel at al. 2001). Some birds also ventured up the west coast of southern Africa as far north as southern Angola; the Benguela upwelling region is highly productive, supporting large numbers of seabirds (Crawford et al. 1991; Shannon and O'Toole 2003).

Both sexes occupied similar foraging areas, with little difference in home ranges between males and females. Catard et al. (2000) did not find any difference in parental trip duration or in meal size between the sexes of white-chinned petrels tracked from the Crozet Islands, suggesting similar parental investment between the sexes. They found that females spent more time searching for prey, whereas males generally made more direct commuting flights to productive areas, but this level of distinction would be hard to detect from the crude GLS location estimates. Berrow et al. (2000b) also reported that breeding males and females from South Georgia foraged in similar areas. Differences in home range or foraging range were not investigated in other studies on white-chinned petrel movements (Weimerskirch et al. 1999; Phillips et al. 2006; Delord et al. 2010a; Péron et al. 2010).

A male bias in fisheries bycatch (particularly longlining) of white-chinned petrels has been found in several studies (Ryan and Boix-Hinzen 1999; Petersen et al. 2009a; Delord et al. 2010b; Chapter 2). This sex bias might result from the larger size of males, which gives them a competitive advantage over the smaller females when foraging for bait and discards from fishing vessels, thus making them more likely to be caught as bycatch (Ryan and Boix-Hinzen 1999). Another possible reason for the sex bias in longline bycatch is a sexual segregation in foraging zones of white-chinned petrels, however Ryan and Boix-Hinzen (1999) found a very strong male bias in white-chinned petrel bycatch from vessels fishing close to breeding islands where sex ratios are assumed to be similar. The lack of sex-linked differences in foraging areas (Berrow et al. 2000b; Catard et al. 2000; this study) suggests that sexual size dimorphism and associated behavioural differences are the most plausible explanation for the sex bias in white-chinned petrel bycatch.

White-chinned petrel foraging ranges were fairly consistent between years both across the small sample of birds tracked, as well as within individuals. It is however important to note that sample sizes of GLS tracked birds during my study were modest (n = 10) and thus some of these differences may have been obscured. My findings are similar to Phillips et al. (2005) who recorded high levels of wintering site fidelity in black-browed albatross from South Georgia. In contrast, Dias et al. (2010) found that individual Cory's shearwaters regularly shift their wintering sites between years, occasionally even alternating between northern and southern hemispheres or between the Indian and Atlantic Oceans.

4.4.1 Effect of breeding and non-breeding stages

Of the 10 white-chinned petrels tracked with GLS loggers, seven bred during the study period. The three birds which did not breed at all probably were pre-breeding individuals, because although c. 15% of adults do not breed every year (Martin et al. 2009), it is unusual for mature birds to have three consecutive breeding sabbaticals. It is thus more likely that these birds were not yet sexually mature. White-chinned petrels are thought to breed for the first time when they are approximately 6 years old (Barbraud et al. 2008), but there is likely to be considerable variation among individuals.

Seabirds experience different foraging constraints during different stages of breeding, with chick-rearing birds under greater energy demands than incubating birds (Shaffer et al. 2003). This typically restricts chick-rearing birds to forage closer to breeding colonies, especially during the early stages when chicks require small meals at regular intervals. Contrary to my results, Berrow et al. (2000b) found that incubating white-chinned petrels from South Georgia ranged more widely than chick-

rearing birds. Results from both GLS and GPS tracks in my study show chick-rearing birds ranging more widely than incubating birds, irrespective of chick age. GPS tracked birds during chick-rearing visited both the South African coast as well as Antarctic waters, however incubating birds only visited the South African coast. Surprisingly, GLS tracks indicate a greater home range size during chick-rearing than during incubation. Similar to my study, Berrow et al. (2000b) and Péron et al. (2010) found that foraging trip durations of white-chinned petrels were longer during incubation than during chick-rearing, although the differences in trip duration between the two breeding periods was greater than my study. The reasons for incubating birds occupying larger foraging ranges than chick-rearing birds, however incubating birds travelling larger distances than chick-rearing birds seem somewhat paradoxical and could perhaps be attributed to small sample sizes or sex imbalance of GLS tracked birds. Birds also covered significantly larger distances per day during incubation than chick-rearing (Berrow et al. 2000b). Delord et al. (2010a) also found that incubating white-chinned petrels from Kerguelen Island made long foraging trips (to Antarctic waters > 60^{o}S), whereas chick-rearing birds alternated shorter foraging trips over the Heard/Kerguelen shelf with longer trips to Antarctic waters. Mean trip duration, total flight distance and maximum range of white-chinned petrels from South Georgia were all lower during chick-rearing than during incubation (Phillips et al. 2006).

White-chinned petrels from South Georgia make wide-ranging trips during the pre-laying exodus (Phillips et al. 2006), and my pre-breeding birds also dispersed more widely than either incubating or early chick-rearing birds, with some reaching the South African coast. Almost all adult white-chinned petrels with enlarged gonads recorded as bycatch off South Africa were caught in the pre-breeding season (September–October; Chapter 2), with very small numbers caught during the incubation and early chick period. The few birds with enlarged gonads caught during the incubation or early-chick periods may have been failed breeders.

Two core areas were utilised by my non-breeding white-chinned petrels; around PEI and into South African waters. All birds foraged off South Africa, while only three birds remained around PEI. The Agulhas Bank and Benguela Current off South Africa are both extremely productive areas that provide rich foraging grounds for a number of seabird species (Shannon and O'Toole 2003). Jackson (1988) found that white-chinned petrels in the southern Benguela fed on a diversity of prey however predominantly on offal from trawlers. In a study investigating non-breeding movements of white-chinned petrels, Phillips et al. (2006) found that birds from South Georgia wintered along the Patagonian Shelf, in similar areas to incubating birds but generally further north. White-chinned petrels from Kerguelen Island (Péron et al. 2010) wintered in similar areas to birds from Marion

Island (current study), which suggests that the entire southwestern Indian Ocean population may winter in the southern African region, however wintering areas of white-chinned petrels from Crozet Islands are unknown.

4.4.2 Overlap with fisheries

Large numbers of seabirds are killed each year by pelagic longline fisheries off South Africa, roughly two-thirds of which are white-chinned petrels (Petersen et al. 2009a; Chapter 2). Prior to the implementation of individual vessel limits for seabird bycatch in the South African pelagic longline fishery (in 2008), up to 1000 white-chinned petrels were killed each year (Petersen et. al 2009a), and in the 1990s an estimated 8000 white-chinned petrels were killed each year by the demersal longline fleet off South Africa (Barnes et al. 1997). Fortunately, numbers killed within the South African EEZ are thought to have reduced considerably in recent years (Petersen et al. 2009b; Chapter 2). There was also some overlap with pelagic longline fisheries off Namibia where some 200 white-chinned petrels are estimated to be killed each year by both pelagic and demersal longline fisheries (Petersen et al. 2007). My data show that adult white-chinned petrels from PEI overlap with pelagic longline fishing effort mainly off South Africa during the non-breeding/winter season. At this time of the year many white-chinned petrels visit South African waters (this study), which coincides with the period when pelagic longline fishing effort is at its highest in South African waters (Chapter 2). The greatest levels of overlap occurred over the productive waters along the edge of the Agulhas Bank, where most seabirds are killed each year (Petersen et al. 2009a; Chapter 2).

There was very little overlap with legal demersal longline fisheries targeting Patagonian toothfish; most areas of overlap were around PEI and further east towards Crozet Island. During the 1990s the toothfish fishery operating around PEI and adjacent islands caught birds at high rates, with white-chinned petrels the most frequently recorded bycatch species, representing c. 80% of bycatch (Nel et al. 2002; Delord et al. 2005; Delord et al. 2010b). White-chinned petrels were killed almost exclusively during their breeding season by toothfish vessels operating close to PEI (Nel et al. 2002). Due to greatly reduced effort and improved seabird mitigation measures, seabird bycatch in the PEI EEZ has declined to negligible levels in recent years, with only three birds (all white-chinned petrels) killed by toothfish fisheries in CCAMLR reporting areas between 2005 and 2014 (CCAMLR 2014). This ignores bycatch from illegal, unreported and unregulated (IUU) toothfish vessels, which is thought to be higher than legal vessels (Nel et al. 2002).

White-chinned petrels are also at risk from fisheries other than longlining; prior to the mid-2000s, demersal trawl fisheries killed large numbers of seabirds, including white-chinned petrels, off South Africa (Watkins et al. 2008). Fortunately, the introduction of additional mitigation measures has greatly decreased seabird bycatch by this fishery (Maree et al. 2014). However, white-chinned petrel mortality may be higher than recorded; the nocturnal habits of the species and difficulties of observing seabird interactions at night may result in the under-recording of this species (Maree et al. 2014).

4.4.3 Conservation implications

To ensure the protection of a species, particularly a wide-ranging seabird species such as the white-chinned petrel, it is important to understand its year-round movements (Lewison et al. 2012). An understanding of seabird movements also is crucial for defining Important Bird Areas (IBAs) and assessing potential interactions with anthropogenic impacts (Lewison et al. 2012). My results confirm that there is little overlap in the at-sea distributions of the two white-chinned petrel subspecies. Most white-chinned petrel populations appear to disperse to different wintering regions (Techow et al. 2016), although there is considerable overlap between southern Indian Ocean populations (this study; Péron et al. 2012). With little or no movement between populations from South Georgia, southern Indian Ocean islands and New Zealand islands these three different populations could be regarded as separate stocks (Ryan et al. 2012). White-chinned petrel bycatch off southern Africa is likely to impact populations from PEI, the Crozets and Kerguelen, which together support a population of ~294 000 breeding pairs (> 1 million fully-grown birds). This population probably can withstand additional mortality of at least 10 000 birds per year (e.g. Barbraud et al. 2008), which is far greater than the current combined bycatch estimates from fisheries off southern Africa and around sub-Antarctic islands.

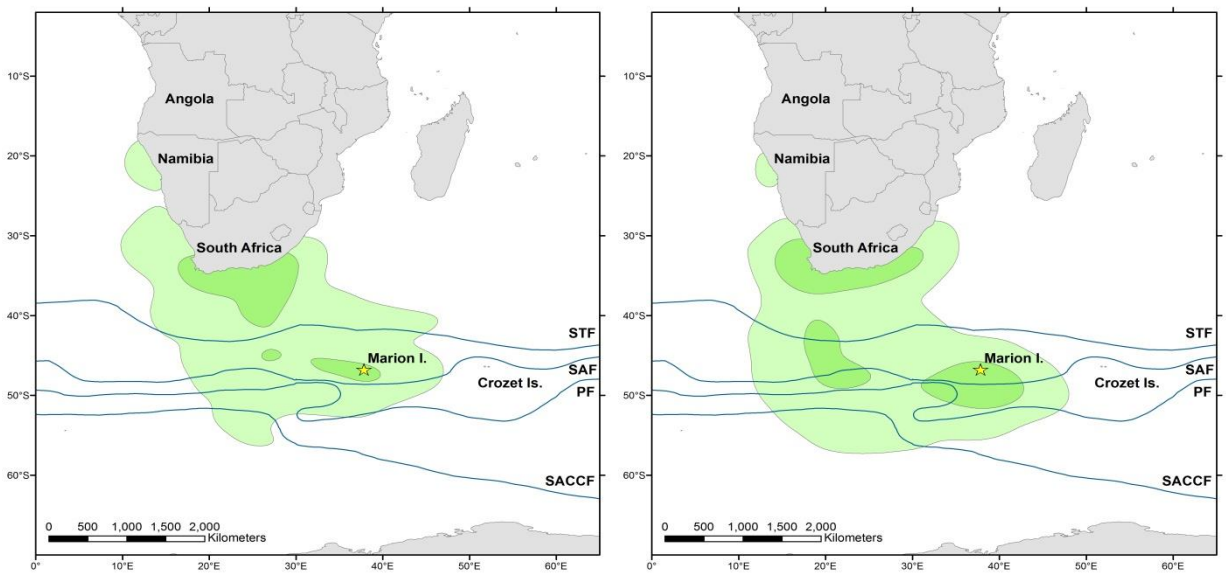
Fishing mortality is not the only threat facing white-chinned petrel populations in the southwestern Indian Ocean region. White-chinned petrels also are impacted on their breeding islands by introduced predators. Rats are significant predators of chicks at breeding sites on Ile de la Possession (Crozet archipelago; Jouventin et al. 2003) and cats kill both adults and chicks on parts of the Kerguelen archipelago (Pontier et al. 2002) and the Crozets (Carboneras et al. 2014). Since the eradication of cats from Marion Island in 1991 the white-chinned petrel population has increased faster than any other burrowing petrel, probably reflecting its greater resilience to attacks by introduced house mice (*Mus musculus*; Dilley et al. 2016). Its increase at Marion suggests that

current fisheries bycatch alone is unlikely to cause decreases in this population of white-chinned petrels. By comparison, the South Georgia population apparently continues to decrease, which is a major concern as it represents > 50% of the global population (Martin et al. 2009). Another potential threat is the effect of climate change and how increasing temperatures might affect seabird species distributions; Krüger et al. (2017) predicted that climate change will cause white-chinned petrels to shift their distributions by almost 10^{o} to the south.

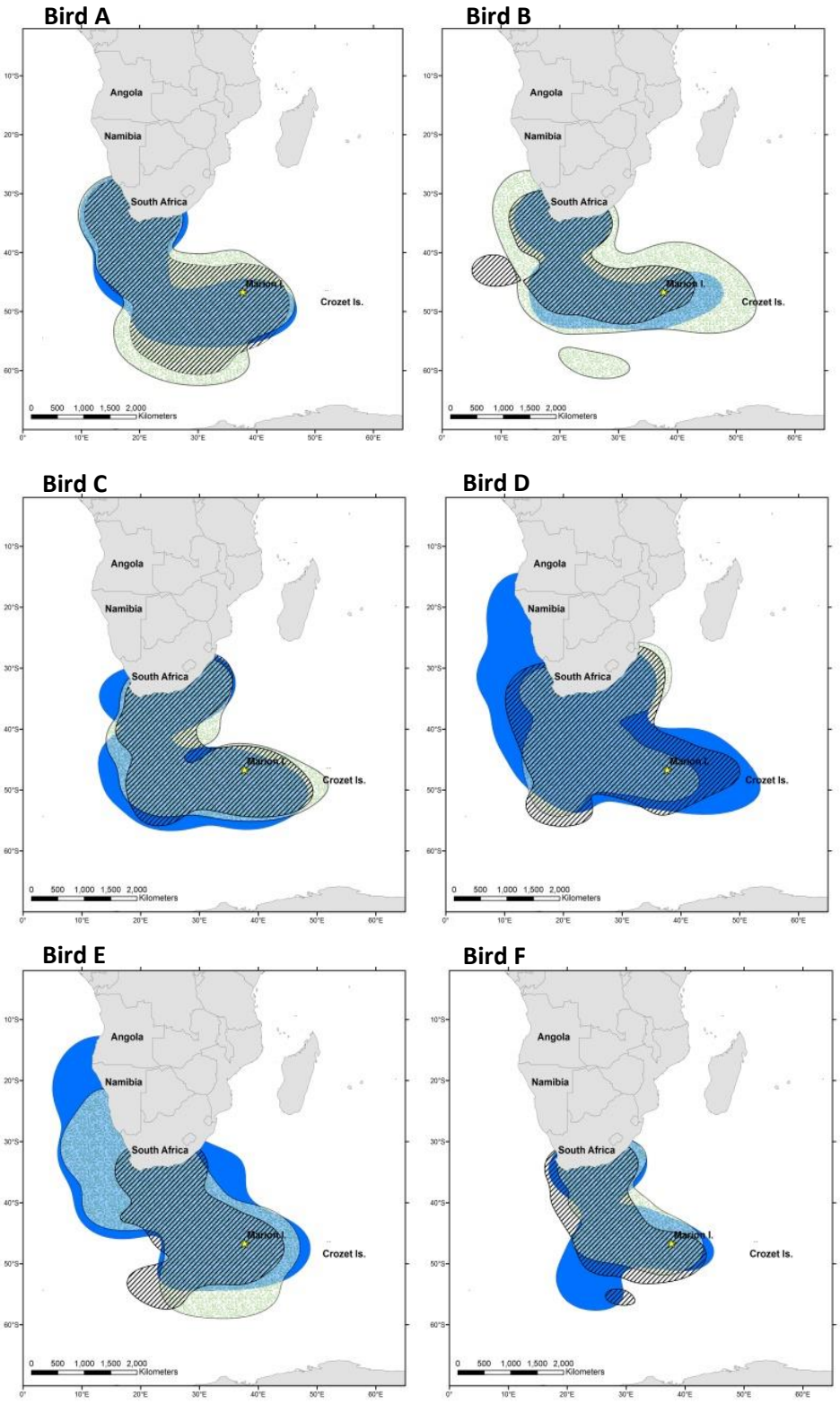
Studies on the at-sea distributions of adult white-chinned petrels from most of its major breeding islands have shed more light on the foraging movements of the different populations. This information is important when considering conservation plans of the species, as well as how different populations may be affected by increased human-induced mortality and potential shifts in prey distribution related to climate change.

Appendix 4.1 Year-round 50% (dark green) and 95% (light green) density kernels of female (a; *n* = 4) and male (b; *n* = 6) white-chinned petrels breeding at Marion Island. ☆ = Breeding colony on Marion Island, STF = Subtropical Front, SAF = Subantarctic Front, PF = Polar Front, and SACCF = Southern Antarctic Circumpolar Current Front.

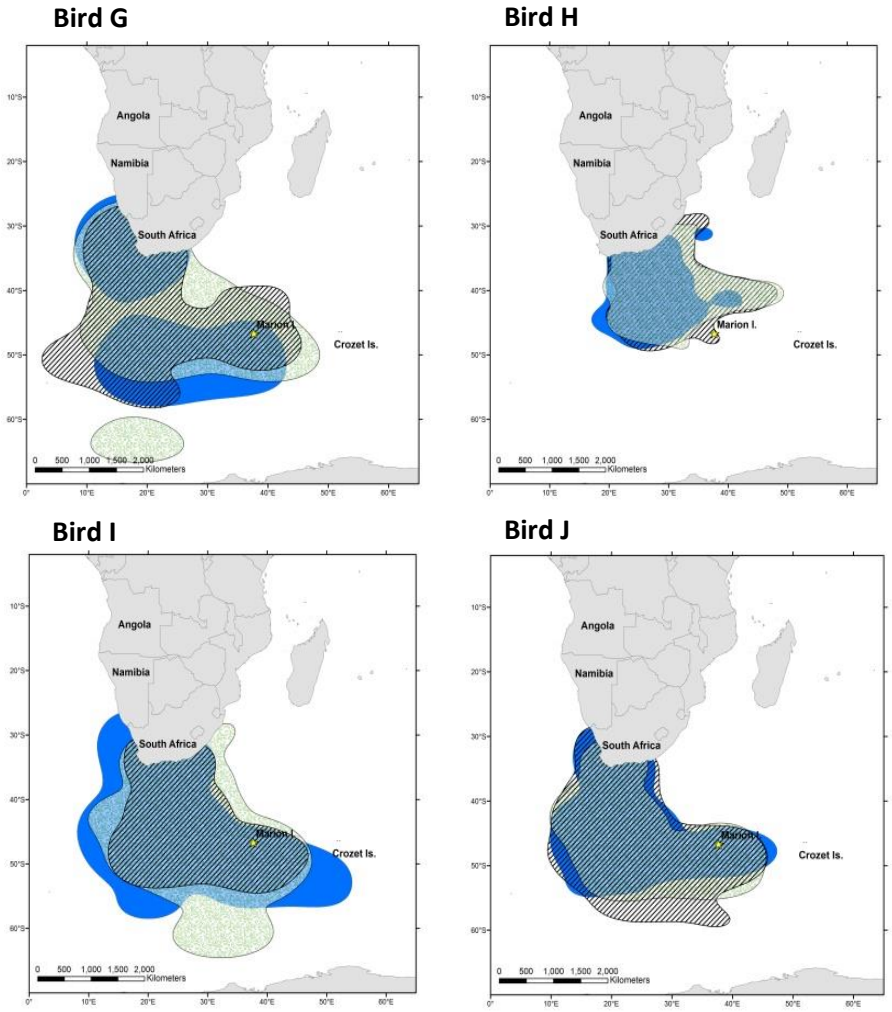
a. **b.**



Appendix 4.2 95% density kernels of white-chinned petrels from Marion Island, during 2009/2010 (green dots), 2011 (diagonal stripes) and 2012 (solid blue). ☆ = Breeding colony on Marion Island.



Appendix 4.2 (continued)



CHAPTER 5

DIVING BEHAVIOUR OF GREY AND WHITE-CHINNED PETRELS AND ITS RELEVANCE FOR MITIGATING LONGLINE BYCATCH

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Abstract

Grey and especially white-chinned petrels are commonly recorded as bycatch in Southern Hemisphere longline fisheries. The white-chinned petrel is in fact the most commonly killed seabird by these fisheries. Despite the importance of understanding seabird diving ability for mitigating longline bycatch, little is known of white-chinned petrel diving behaviour, with no previous studies on the diving behaviour of grey petrels. To investigate their diving behaviour I obtained data from TDRs from nine white-chinned petrels breeding on Marion Island, southwestern Indian Ocean, during the late incubation and chick-rearing period, and seven grey petrels breeding on Gough Island, South Atlantic Ocean, during late incubation. Maximum dive depth of white-chinned petrels (16 m) was slightly deeper than the previous estimate (13 m), with grey petrels diving considerably deeper (22 m). Maximum dive duration was 22 s for white-chinned petrels and 39 s for grey petrels. Maximum dive depth and duration varied considerably among individuals for both species. Male white-chinned petrels dived deeper than females and birds feeding chicks dived deeper than incubating birds, but dive rate did not differ between the sexes. The time of day influenced dive depth for grey petrels; day dives were on average deeper than night dives, but the effect was weak. By providing insight into the diving behaviour of these two *Procellaria* petrels my findings help to explain their high mortality on longlines. I suggest that bird-scaring lines provide protection while longlines are being set to a depth of at least 10 m, which could be achieved by adding weight to branchlines, reducing vessel setting speeds and redesigning bird-scaring lines.

5.1 Introduction

Thousands of seabirds are killed each year by longlines whilst diving for baited hooks. Despite this seabird mortality from longline fisheries, both demersal and pelagic, has decreased in recent years, mostly through the use of effective mitigation measures which have been devised to protect baited hooks until they sink below the birds' diving range. The most effective mitigation measures include bird-scaring lines, adding weight to the hooks to increase line sink rates, adjusting line setting speeds and also the timing of line setting (Robertson 2000). To successfully reduce seabird bycatch, it is important to understand when, where and how deep bycatch-susceptible seabird species can dive.

White-chinned petrels are the most common seabird species caught on longline fishing gear in the Southern Ocean (Delord et al. 2005, 2010b; Moreno et al. 2006; Robertson et al. 2006; Petersen et al. 2009b; Ryan et al. 2012) while grey petrels are also commonly caught by longliners (Bell et al. 2013). Both species regularly scavenge at fishing vessels (Robertson 2000), where they can facilitate

the mortality of larger, shallower-diving species, such as albatrosses. By returning baited hooks to the surface from beyond the diving capabilities of the larger species, the deeper-diving species allow the larger species a second chance at obtaining baited hooks; on returning to the surface, the deeper diving species are regularly out-competed for the baited hook by the larger species, resulting in the potential mortality of the larger species (Jiménez et al. 2012).

White-chinned petrels disperse widely throughout the Southern Ocean during the breeding season with birds foraging from the subtropics to the pack ice (Weimerskirch et al. 1999; Chapter 4). Non-breeding birds range into the tropics, exposing them to a wide range of longline fisheries (Chapter 4). Despite a worldwide population comprising more than one million pairs (Ryan et al. 2012), white-chinned petrels are listed as Vulnerable by the IUCN because of the high levels of bycatch and evidence that some populations are decreasing (IUCN 2016). Grey petrels have a circumpolar distribution in sub-Antarctic and temperate waters, mostly between 32–58^{o}S (ACAP 2009c). They breed on a number of sub-Antarctic islands; Macquarie, Amsterdam, Crozet, Kerguelen, Campbell, Antipodes, PEI, Gough and possibly Tristan da Cunha, with the largest populations occurring on the Antipodes (32 000–73 000 pairs, Bell et al. 2013) and Gough Island (10 000 pairs; Cuthbert and Sommer 2004). On a number of these islands breeding success has been reduced by introduced predators, such as house mice and rats (Cuthbert and Hilton 2004; Schulz et al. 2006; Dilley et al. 2015). The high levels of bycatch from fisheries, together with predation from introduced predators on their breeding islands, have led to the grey petrel being listed as Near-Threatened (IUCN 2016).

The nocturnal diving ability of both grey and white-chinned petrels is of particular interest because night-time setting of longlines is one of the key mitigation measures to reduce seabird bycatch in Southern Ocean fisheries (Murray et al. 1993; Cherel et al. 1996). However it is believed that both species gain little benefit from restricting line setting to the night because they appear to be equally proficient at foraging at night as during the day (Harper 1987; Delord et al. 2010b). The theory of high nocturnal activity of both grey and white-chinned petrels is further supported by the analysis of bycatch data of both species, where night-time bycatch rates are either similar or higher than daytime sets (Barnes et al. 1997; Nel et al. 2002; Delord et al. 2010b; Mackley et al. 2011). White-chinned petrel foraging activity apparently increases around full moon, but this has only been demonstrated during the non-breeding season (Mackley et al. 2011), while grey petrel longline bycatch is known to increase during periods of high lunar luminance (Delord et al. 2010b).

Currently, little is known of *Procellaria* petrel diving behaviour. The maximum dive depths of white-chinned and Westland petrels (*P. westlandica*) has been measured with the use of capillary

depth gauges (Huin 1994; Freeman et al. 1997). Unfortunately, these depth gauges give no indication when the deepest dive takes place, or dive velocity in the water column. Bell (2016) has recently investigated the diving behaviour of black petrels (*P. parkinsoni*) with the use of depth recorders (TDRs); this is the only previous study of *Procellaria* petrel diving behaviour using TDRs. Seabird diving data are now available for a range of petrel and shearwater species (Weimerskirch and Sagar 1996; Burger 2001; Ronconi et al. 2010). There have been no previous studies investigating the diving behaviour of grey or white-chinned petrels using TDRs, thus highlighting the importance of my study.

5.2 Methods

The diving ecology of grey and white-chinned petrels was investigated using birds caught in their breeding burrows on Gough Island (40°21'S, 09°53'W), central South Atlantic Ocean and Marion Island (46°52'S, 37°51'E) in the Southern Ocean, respectively. TDRs (G5; Cefas Technology Limited; UK; 35.5 x 11.5 mm; 2 Mb memory; mass 2.7 g) were programmed to record temperature and pressure data every 2 s for white-chinned petrels and pressure data every 3 s for grey petrels. The relatively course sampling frequency was chosen to ensure as many days as possible were included in the dive profile of the birds' foraging trips. Some white-chinned petrels and all grey petrels were also fitted with a GPS device which was employed in tandem with the TDR devices, housed together in a heat shrink tube, making them both streamlined and waterproof. Movement data from the white-chinned petrels are analysed in Chapter 4 with grey petrel movement data to be analysed in another study. The combined TDR/GPS capsule (31 mm wide; 59 mm long; 13 mm high; ~15% of the cross sectional width of the bird) weighed 27.7 g (< 3% of body mass; Barron et al. 2010). The device was taped to the birds' back feathers using Tesa tape and the feathers in front were combed over the device to cover ~50% of the device. Devices of similar size with similar attachment methods have been used on other procellariiform species, without any measureable negative effects (Delord et al. 2010a). Handling lasted < 5 min, with weighing and measuring done on device retrieval. Birds were sexed by measuring their culmen length and bill depth at the gonys to the nearest 0.1 mm with Vernier callipers (Ryan 1999).

TDRs were deployed on six white-chinned petrels (five females, one male) during the late incubation period (18 November–05 December 2012) and six (two females, four males) during the chick rearing period (01 January–06 February 2013). TDRs were only deployed during the mid-incubation period (06 April 2014) for grey petrels, which included three on females and five on

males. Nest burrows were checked regularly thereafter, via inspection hatches until the equipped birds returned, or were trapped in the burrow by fitting a one-way flap to burrow entrances.

TDR data were analysed using IGOR Pro (Wavemetrics Inc.; USA; 2008; Version 6) and were corrected for surface drift (the slight inaccuracy in the calculation of the water surface). To exclude noise in the data, we restricted analyses to dives < 1 m, while for those dives with substantial baseline variation, dives < 2 m were excluded. The relatively coarse sampling interval (white-chinned petrels = 2 s; grey petrels = 3 s) reduced my ability to discriminate short dives. Maximum dive depths and dive durations were recorded, although both estimates were conservative due to the coarse sampling intervals, which also effected the calculation of descent and ascent rates; thus only dives > 5 m deep were used for this calculation. Dives were categorised as either U- or V-shaped, with shape identified by the presence or absence of a bottom-phase at or near the maximum dive depth, when the dive rate was < 0.25ms^{-1} (Kato et al. 2006). U-shaped dives were most likely those dives where prey was actively pursued (Shepard et al. 2010) while during V-shaped dives prey was simply seized before returning to the surface.

I estimated dive frequency in relation to diel cycles and moon phase. Only dives with associated GPS co-ordinates or for which time of day could be confidently ascribed to day or night (with use of TDR temperature data to determine location) were used to test the influence of time of day and lunar effects. Sunrise/set times as well as moonrise/set times were calculated based on GPS coordinates. As no incubating white-chinned petrels carried GPS devices, their positions could not be precisely determined, however approximate latitudinal location was inferred from temperature data and thus all dives could be confidently ascribed to day or night. Unfortunately TDRs deployed on grey petrels did not record temperature and as GPS/TDR overlap did not equal 100%, only 88% of recorded dives could be ascribed to day or night. Moon phase was placed into three equal categories based on percentage illumination (0–33%, 34–66% and 67–100%) for white-chinned petrels, while it was recorded as present or absent for grey petrels. Unfortunately I was unable to assess the effect of cloud cover on lunar illumination.

All analyses were performed in the R statistical package (R Core Team 2015). Values given are means ± standard deviation, unless otherwise stated. Generalised linear mixed models (GLMM) with a Gaussian (white-chinned petrels) and Gamma distribution (grey petrels) using a logarithmic link function were used to determine the influence of various explanatory variables (sex, time of day and lunar effects) on dive duration and dive depth, with individual bird set as the random effect (Nakagawa and Shielzeth 2010). These models were created using the NLME package (Linear and Nonlinear Mixed Effect Models). Different models were used to test the influence of time of day and

lunar effects. Models testing time of day used all dives which could be ascribed to day or night (white-chinned petrel = 100%; grey petrel = 88%). Models which incorporated lunar luminance for white-chinned petrels, only included those dives undertaken during the night, after moonrise and before moonset (*n* = 47), while lunar models for grey petrels used all night dives (*n* = 302) to test the effect of moon presence/absence. AIC and model averaging were used to select the most parsimonious models (Appendix 5.1 and 5.2). To relate dive duration to dive depth I tested both linear and exponential models and selected the model which gave the best coefficient of determination (r^2). T-tests and Analysis of Variance (ANOVA) were used to test whether dive rates differed between sexes and stages of breeding.

5.3 Results

Loggers were recovered from 10 of the 12 white-chinned petrels, but data were only retrieved from nine TDRs (four males; five females): five from incubators and four from birds provisioning chicks. Eight grey petrels were recaptured carrying both GPS and TDR devices but data were retrieved from only seven of these paired devices (four males, three females). TDR data were sampled for 9.2 ± 3.1 d (range 5–13 d) and 15.2 ± 4.1 days at sea (range 7–19 days) for white-chinned and grey petrels, respectively (Table 5.1). There was large variation in number of dives per individual for both species (white-chinned petrel range 2–76; grey petrel range 3–395).

Table 5.1. Summary of device coverage of white-chinned (WCP) and grey petrels (GP).

Bird ID (sex)	Body mass (g) at recovery	Device	At-sea TDR (days)	At-sea GPS (days)	TDR/GPS overlap (%)
White-chinned petrels	-	-	-	-	-
Incubating	-	-	-	-	-
A (F)	1180	TDR only	13*	-	-
B (F)	1220	TDR only	9*	-	-
C (F)	1290	TDR only	10*	-	-
D (M)	1510	TDR only	5*	-	-
E (F)	1360	TDR only	6*	-	-
Average ± SD	1312 ± 130	-	8.6 ± 3.2	-	-
Chick rearing	-	-	-	-	-
F (M)	1210	TDR & GPS	9	23	100
G (F)	1070	TDR & GPS	13	26	100
H (M)	1120	TDR & GPS	11	33	100
I (M)	1080	TDR & GPS	8	24	100
Average ± SD	1120 ± 63	-	10.3 ± 2.2	26.4 ± 4.7	100
WCP average ± SD	1227 ± 142	-	9.2 ± 3.1	26.4 ± 4.7	100
Grey petrels	-	-	-	-	-
A (F)	1090	TDR & GPS	19	12	64.0
B (M)	1050	TDR & GPS	17	15	87.1
C (M)	1070	TDR & GPS	17	8	44.9
D (F)	1040	TDR & GPS	14	14	98.6
E (M)	1010	TDR & GPS	14	4	29.3
F (F)	1120	TDR & GPS	18	5	30.3
G (M)	1230	TDR & GPS	7	14	100
GP average ± SD	1147 ±	-	15.2 ± 4.1	10.3 ± 4.4	67.7 ± 30.9

* At-sea TDR days for incubating white-chinned petrels were estimated based on dive and device deployment/retrieval data.

5.3.1 Maximum dive depth and duration

Maximum dive depth varied considerably among individuals of both species (white-chinned petrel range 2.1–16.1 m; grey petrel range 2.4–21.9 m), with white-chinned petrels averaging shallower dives (2.9 ± 2.4 m) than grey petrels (3.2 ± 2.2 m; Table 5.2). There was also large individual variation in maximum dive duration between both species with white-chinned petrels averaging shorter dives (4.6 ± 3.9 s; range 2–22 s) than grey petrels (7.7 ± 5.6 s; range 6–39 s; Table 5.2). Most dives made by white-chinned petrels were < 5 m deep (88%), with 95% of dives < 7 m deep (Fig. 5.1). Almost two

thirds (62%) of grey petrel dives were <5 m and 95% of dives were <7 m (Fig. 5.1). White-chinned petrel maximum dive depth was related to number of dives sampled according to: dive depth (m) = 2.57 * ln (number of dives) + 0.806 (r^2 = 0.42), with a similar equation for grey petrels: maximum depth (m) = 2.3575 x ln (number of dives) + 1.8486 (r^2 = 0.4766). Dive duration for white-chinned petrels was related to dive depth: dive duration (s) = 0.548 * dive depth (m) + 0.528 (r^2 = 0.63), with a similar equation representing grey petrels: dive duration (s) = 2.0886 x dive depth (m) + 0.787 (r^2 = 0.6964).

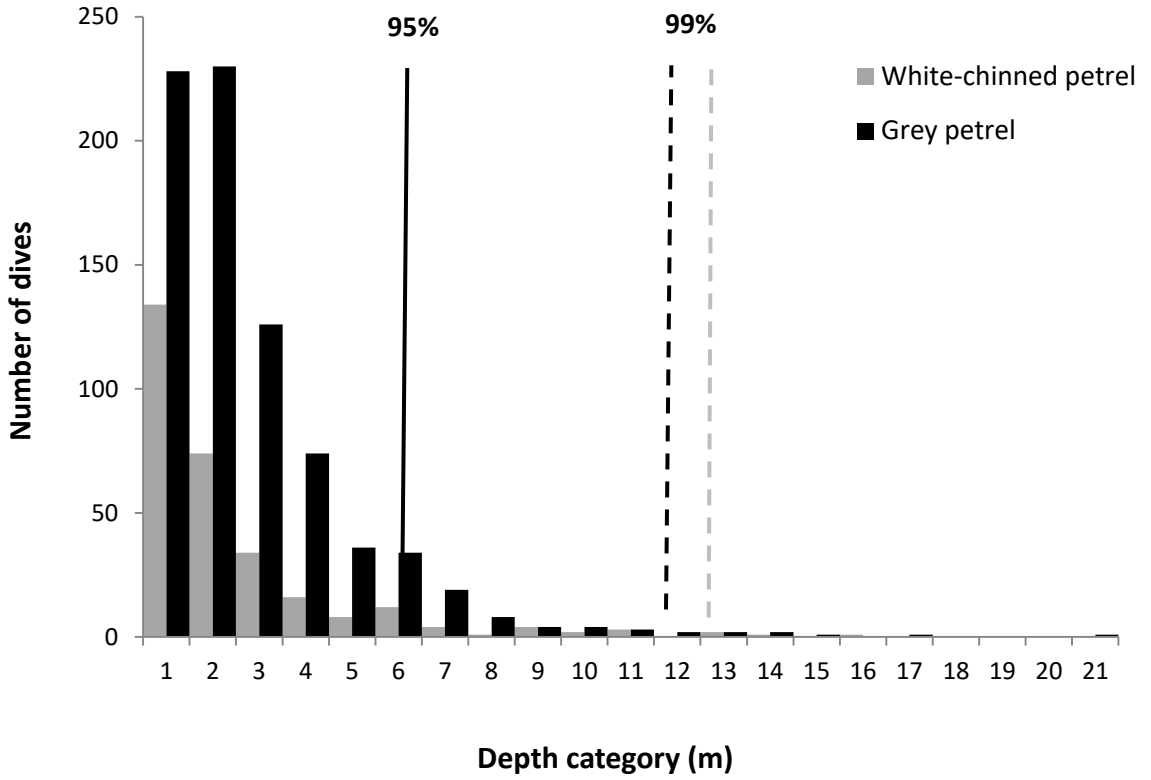


Fig. 5.1. Number of dives per depth category, for white-chinned and grey petrels, with 95 and 99% of dives indicated.

5.3.2 Dive characteristics

Dive ascent rate (white-chinned petrels 1.6 ± 1.1 m.s $^{-1}$; grey petrels 1.2 ± 0.7 m.s $^{-1}$) averaged faster than dive descent rate (white-chinned petrels 1.5 ± 0.9 m.s $^{-1}$; grey petrels 1.1 ± 0.5 m.s $^{-1}$). Ascent and descent rates for white-chinned petrel dives < 5 m deep were slower (1.5 ± 0.9 m.s $^{-1}$; 1.4 ± 0.9 m.s $^{-1}$) than dives > 5 m deep (1.9 ± 1.4 m.s $^{-1}$; 1.7 ± 0.9 m.s $^{-1}$), while the ascent and descent rates were similar for grey petrel dives > 5 m deep (1.2 ± 0.5 m.s $^{-1}$; 0.9 ± 0.3 m.s $^{-1}$) compared to dives < 5 m deep (1.3 ± 0.7 m.s $^{-1}$; 1.1 ± 0.6 m.s $^{-1}$). V-shaped dives accounted for 89 and 90% of the dives for

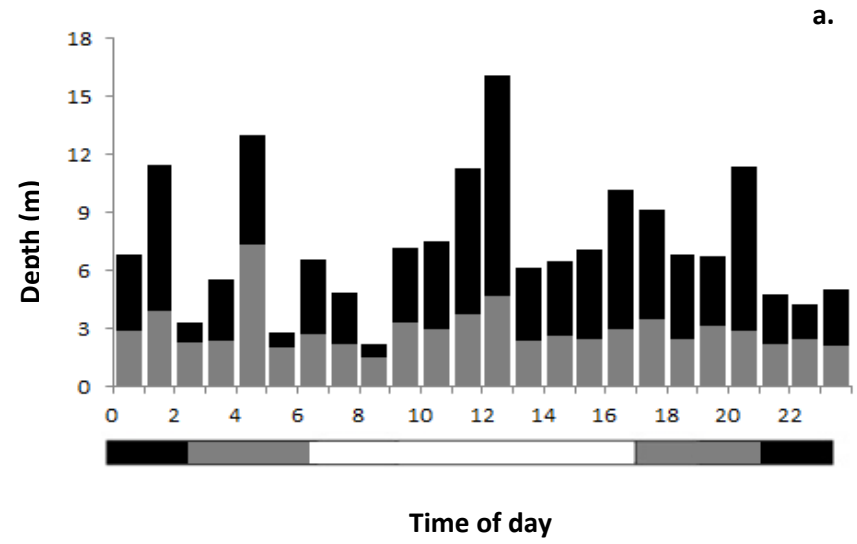
white-chinned and grey petrels, respectively (Table 5.2). White-chinned petrel dive type (U- or V-shaped) had no effect on dive depth (t = 0.44; p = 0.66), but U-shaped dives (7.4 ± 4.5 s) lasted longer than V-shaped dives (4.3 ± 3.6 s; t = 5.51; p < 0.001). Dive type affected both dive duration and depth for grey petrels, with U-shaped dives lasting longer (14.2 ± 7.2 s) and deeper (4.4 ± 2.6 m) than V-shaped dives (6.9 ± 4.9 s; t = 1.65; p < 0.001, 3.2 ± 2.2 m; t = 1.65; p < 0.001). There was no significant difference in the frequency of U- and V-shaped dives between day and night for either species.

Table 5.2. Summary of diving activity of white-chinned and grey petrels. Dives without GPS co-ordinates or for which day or night could not be confidently assigned were not used to calculate percentage of night dives.

Bird ID (sex)	**Total dives**	**Night dives (%)**	**V-shaped (%)**	**Dive duration (s)**		**Dive depth (m)**	
				Max	**Mean ± SD**	**Max**	**Mean ± SD**
White-chinned petrel	-	-	-	-	-	-	-
A (F)	2	100	100	2	2.0 ± 0.0	2.1	1.7 ± 0.4
B (F)	61	45.9	83.6	22	5.3 ± 3.9	6.8	2.6 ± 1.4
C (F)	76	21.1	89.5	22	3.8 ± 3.2	13.4	2.1 ± 1.7
E (F)	17	17.6	82.4	16	6.3 ± 3.3	6.5	2.8 ± 1.6
G (F)	23	95.7	95.7	14	4.4 ± 3.5	10.2	3.5 ± 2.9
All females	179	39.7	87.7	22	4.4 ± 3.6	13.4	2.4 ± 1.7
D (M)	14	35.7	92.9	20	6.6 ± 6.0	7.5	2.9 ± 2.2
F (M)	31	67.7	90.3	20	6.3 ± 5.3	16.1	4.6 ± 3.9
H (M)	40	15	90.0	6	3.5 ± 1.6	6.1	2.4 ± 1.4
I (M)	32	21.9	87.5	10	3.1 ± 1.8	11.4	3.1 ± 2.2
All males	117	33.3	89.7	20	5.0 ± 4.3	16.1	3.6 ± 3.0
All white-chinned petrels	296	37.2	88.5	22	4.7 ± 3.9	16.1	2.9 ± 2.4
Grey petrel	-	-	-	-	-	-	-
A (F)	8	88	75	30	11.3 ± 9.1	5.8	2.4 ± 1.5
D (F)	89	24	88	39	10.0 ± 8.0	21.9	4.8 ± 3.7
F (F)	47	98	85	27	7.2 ± 5.1	8.4	2.4 ± 1.4
All females	144	51	86	39	9.2 ± 7.3	21.9	3.9 ± 3.2
E (M)	6	67	100	12	5.5 ± 4.0	7.2	2.6 ± 2.3
B (M)	395	30	93	18	6.0 ± 3.2	10.5	2.8 ± 1.3
C (M)	227	77	86	36	9.6 ± 6.9	15.0	3.7 ± 2.6
G (M)	3	67	100	6	4.0 ± 1.7	2.4	1.6 ± 0.7
All males	631	44	91	36	7.3 ± 5.1	15.0	3.1 ± 1.9
All grey petrels	775	44	90	39	7.7 ± 5.6	21.9	3.2 ± 2.2

5.3.3 Effect of time of day

Time of day had a minor influence on white-chinned petrel average maximum dive depth with a greater influence on average maximum dive duration; birds made slightly longer dives during the day (4.9 ± 4.2 m) than at night (4.2 ± 3.4 m; Table 5.1 and 5.2), however none of these differences were significant (Appendix 5.2a). On average grey petrels dived 10% deeper during the day (3.4 ± 2.3 m) than during the night (2.9 ± 1.9 m). There was a difference in grey petrel dive duration between day (7.5 ± 5.4 s), and night dives (7.4 ± 5.5 s; Fig. 5.2 and 5.3). For both species more dives took place during the day (white-chinned petrel 56%; grey petrel 63%) than at night (white-chinned petrel 44%; grey petrel 37%; Table 5.3), despite this there was no significant difference in dive rate between day or night, as daylight hours (%) were similar to the dive percentages.



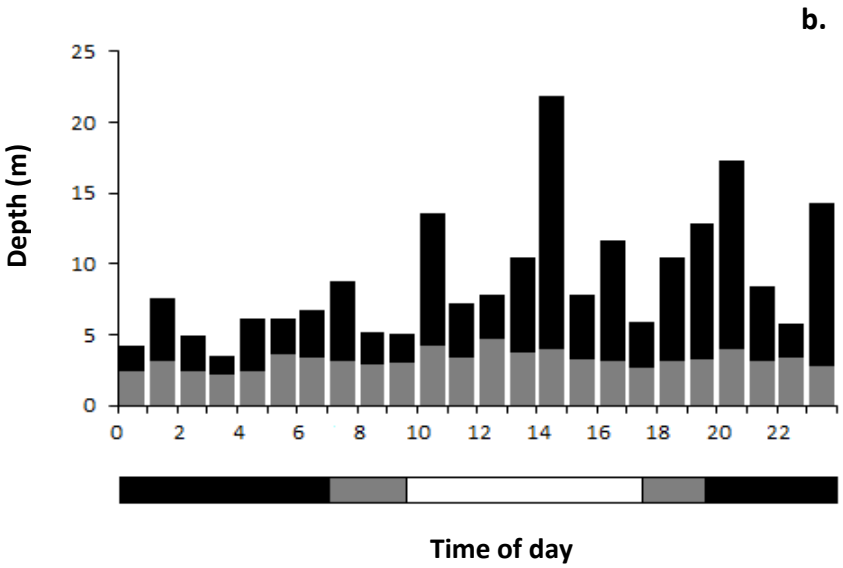


Fig. 5.2. Maximum (black bars) and average (grey bars) dive depth for a. white-chinned and b. grey petrels according to time of day. Scale bar beneath represents approximate light levels throughout the day; black night, white day and grey unclear (due to differences in positions of the birds).

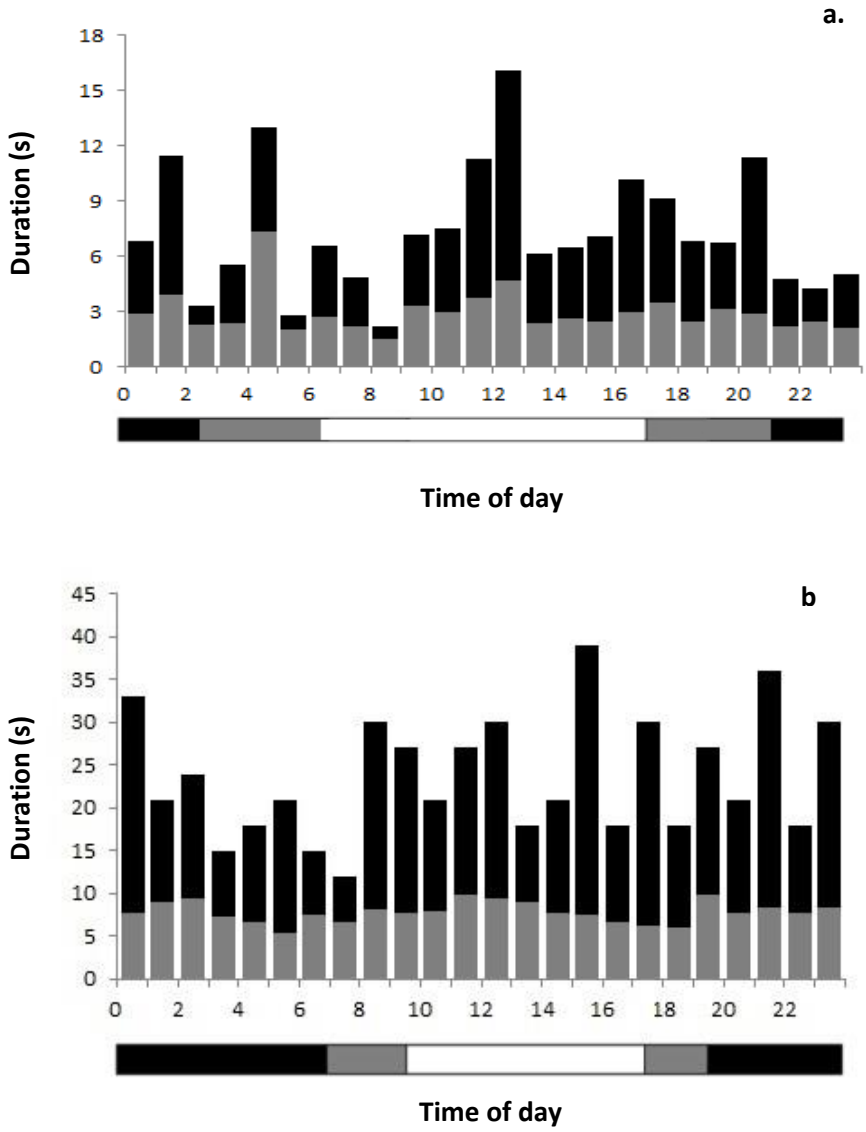


Fig. 5.3. Maximum (black bars) and average (grey bars) dive duration for a. white-chinned and b. grey petrels according to time of day. Scale bar beneath represents approximate light levels throughout the day; black night, white day and grey unclear (due to differences in positions of the birds).

5.3.4 Effect of lunar phase

Moon phase had little influence on white-chinned petrel dive duration, dive depth or dive rate (but statistical power was low due to the small total number of night dives). Grey petrel night dives with

the moon present averaged shorter and shallower (7.1 ± 5.2 s; 2.6 ± 1.5 m) than night dives with the moon absent (8.1 ± 6.2 s; 3.6 ± 2.7 m). Despite this the presence/absence of the moon was excluded from the best model predicting dive duration, with no suitable model selected for dive depth (Appendix 5.1b).

5.3.5 Effect of sex

White-chinned petrel dive duration differed between the sexes, with males diving longer (5.0 ± 4.3 s) than females (4.4 ± 3.6 s; Appendix 5.2a). However, most white-chinned petrels sampled during incubation were female (4 of 5 birds), whereas most birds sampled during chick rearing were male (3 of 4 birds). Female grey petrels dived on average deeper (3.9 ± 3.2 m) and longer (9.2 ± 7.3 s) than males (3.2 ± 1.9 m; 7.3 ± 5.1 s). However sex did not have any significant effect on grey petrel dive depth or duration (Appendix 5.1b and 5.2b). At night, females dived somewhat longer than males (Appendix 5.2b). There was no significant difference in dive rate between male (0.09 dives h^{-1}, n = 117) or female (0.16 dives h^{-1}; n = 179; t = 0.85; p = 0.210) white-chinned petrels, nor between grey petrel sexes; females (0.1 dives h^{-1}; n = 144), males (0.4 dives h^{-1}; n = 631; t_{775} = 1.860; p = 0.28).

5.3.6 Stage of breeding

Although none of the variables tested had particularly strong effects on white-chinned petrel dive depth or duration (when either time of day or lunar luminance was included), stage of breeding had the most influence on both dive depth and dive duration, with birds provisioning chicks diving deeper (3.6 ± 2.9 m) but shorter (4.5 ± 3.8 s) than incubating adults (2.4 ± 1.6 m; 4.8 ± 4.0 s; Appendix 5.1a). Grey petrel diving behaviour was investigated during late incubation only and thus the effect of breeding stage could not be determined for this species.

5.4 Discussion

5.4.1 Maximum dive depth and duration

Despite the limitations of my study (small sample size, low TDR sampling frequency and the relatively large TDR/GPS package) both white-chinned and grey petrels dived to considerable depths. Both species dived beyond the maximum dive depth as recorded by Freeman et al. (1997) investigating Westland petrel dive depth (average maximum depth 3.2 ± 3.8 m; maximum depth 7.6

m), or Huin (1994) investigating white-chinned petrels dive depth (average maximum depth 6.2 ± 3.0 m; maximum depth 12.8 m) both using depth gauges. With the use of TDRs, Bell (2016) recorded black petrels diving to a depth of 34.3 m, averaging 2.6 ± 0.1 m. The small size and relatively cheap prices of TDR devices means that the diving behaviour of many seabird species can now be accurately investigated, even those species which were previously considered too small to handle bulky recording devices.

Procellaria petrels generally do not dive as deep as *Puffinus*/*Ardenna* shearwaters, which have been recorded to depths of over 70 m (Weimerskirch and Sagar 1996; Weimerskirch and Cherel 1998), however *Procellaria* petrels (particularly white-chinned petrels) are regularly recorded as incidental bycatch on longlines operating off South Africa (Petersen et al. 2009b; Chapter 2). It is thus surprising that many more deep-diving *Puffinus*/*Ardenna* shearwaters are not recorded as pelagic longline bycatch off South Africa. The lack of the deep-diving shearwater bycatch may be explained by their low numbers associated with longline vessels off South Africa (Chapter 3). Despite the greater diving abilities of grey petrels compared to white-chinned petrels (current study), grey petrels are very rarely recorded as pelagic longline bycatch off South Africa. Analysis of seabird attendance data associated with longline vessels reveals that grey petrels do not attend pelagic longline vessels in large numbers (Chapter 3) and thus bycatch figures are significantly lower than white-chinned petrels (Chapter 2).

5.4.2 Dive characteristics

U-shaped dives were generally deeper and longer than V-shaped dives. This is as U-shaped dives are thought to be those dives when birds are actively pursuing prey at depth (Shepard et al. 2010), while V-shaped dives are shorter, shallower dives, perhaps undertaken whilst scavenging for fisheries discards behind fishing vessels. As longline fleets regularly operate in areas where both white-chinned and grey petrels are known to forage (ACAP 2009c; ACAP 2012; Chapter 4), it is possible that fishery interactions occurred and that the many V-shaped dives were made whilst scavenging behind these vessels.

5.4.3 Nocturnal activity

White-chinned petrels are highly nocturnally active and regularly forage around fishing vessels during the day and night (Péron et al. 2010; Mackley et al. 2011). There have been no previous

foraging activity studies on grey petrels, however the large numbers of grey petrels caught on longlines set at night, suggests they too are nocturnally active (Delord et al. 2005, 2010b). Dive rates in my study were similar between day and night, further highlighting the nocturnal activity of these two *Procellaria* petrels. The greater availability to prey at night is perhaps the reason these two species are nocturnally active (Péron et al. 2010), as small pelagic fish are known to move closer to the surface at night (Beckley and van der Lingen 1999). Both species dived slightly deeper during the day than at night presumably linked to greater light levels or because prey were deeper during the day (Wilson et al. 1993). Although the setting of longlines at night has significantly reduced the bycatch of many species from longline fisheries (Delord et al. 2005), results from my study further confirm that night setting alone will not effectively reduce *Procellaria* petrel bycatch, and that additional mitigation measures need to be used along with night setting.

5.4.4 Lunar influence

Moon phase affects light levels and is known to affect seabird bycatch from longline fisheries (Klaer and Polachek 2001; Petersen et al. 2009b; Delord et al. 2010b). Birds are more frequently caught over full moon than over new moon periods (Petersen et al. 2009b; Chapter 2), likely due to increased light levels, resulting in the increased foraging activity of many seabirds (Harrison et al. 1991; Hedd et al. 2001). The lack of lunar influence on dive depth or dive duration in this study may be due to the small sample sizes or because both studies only encompassed one complete lunar cycle each. Moon phase is known to influence the foraging activity of many other Procellariiformes (Jouventin and Weimerskirch 1990; Mackley et al. 2011; Pinet et al. 2011) and bycatch on longlines is known to increase during periods of high lunar luminance for both white-chinned (Petersen et al. 2009b; Chapter 2) and grey petrels (Delord et al. 2010b). Introducing stricter seabird bycatch mitigation measures during periods of high seabird bycatch, such as the couple days surrounding full moon, will help to further reduce seabird bycatch from pelagic longline fisheries, without significantly impacting on fishing operations.

5.4.5 Effect of sex

Male white-chinned petrels dived deeper and for longer periods than females, while sex did not have a strong influence on grey petrel dive depth or duration, but females tended to dive deeper than males. Skewed sex ratios in seabird bycatch have frequently been recorded for *Procellaria* petrels (Murray et al. 1993; Nel et al. 2002; Petersen et al. 2009b; Bugoni et al. 2011). Bycatch

studies have regularly found greater proportions of male white-chinned petrels (Ryan and Boix-Hinzen 1999; Nel et al. 2002; Petersen et al. 2009b), which could be explained by the greater body size and hence superior diving ability of males. The sex imbalance between incubating and chick-rearing birds may however have also played a role in determining dive differences between the sexes. Contrary to this, several longline bycatch studies have recorded greater proportions of female grey petrels than males (Bartle 1990; Murray et al. 1993; Gales et al. 1998). This sex bias has been attributed to sexual segregation of foraging zones of grey petrels (Bartle 1990), which was not apparent in my year-round movement study of white-chinned petrels (Chapter 4). A sex bias in seabird bycatch is concerning as seabirds require both species to successfully raise chicks (Delord et al. 2005).

5.4.6 Breeding stage

White-chinned petrels dived deeper during chick provisioning than during incubation, perhaps because chick rearing birds are under greater energy demands to provide for chicks, while incubating birds forage to maintain body condition only (Schaffer et al. 2003). Foraging trip durations of incubating white-chinned petrels were longer than foraging trip durations during chick-rearing (Phillips et al. 2006; Berrow et al. 2000b; Delord et al. 2010a; Péron et al. 2010; Chapter 4), presumable as chick-rearing birds need to provide meals to chicks at regular periods. Incubating white-chinned petrels from PEI occupied a similar core area to early chick-rearing birds, however early chick-rearing birds occupied a greater home range during this period (Chapter 4). My study investigated the diving behaviour of grey petrels during late incubation only, and thus the effect of breeding stage was not investigated for this species.

5.4.8 Vulnerability to seabird bycatch

Scavenging seabirds with deep-diving abilities are more susceptible to mortality from longline bycatch. This is because deeper-diving species are able to access sinking hooks for longer periods of time than shallow-diving species. Like many other seabird species, *Procellaria* petrels are attracted to longline vessels to take advantage of the large quantity of bait and offal discards (Chapter 3). Dive behaviour behind fishing vessels is likely to be altered due to the abundance of food and it is perhaps whilst scavenging from fishing vessels that many of the shallow dives are undertaken (Gremillet et al. 2008), unfortunately this theory could not be confirmed in my study. This study has shown that both *Procellaria* petrels are highly active at the surface, with 87% (white-chinned petrels)

and 85% (grey petrels) of recorded dives < 5 m deep. Further to this, some birds undertook very few detectable dives throughout their entire foraging trip and thus it is assumed many shallow dives (< 1 m deep) were not detected by the TDR devices.

Despite this, both species have the ability to dive relatively deep, which together with their nocturnal activity, explains their high mortality from longlines. Dive descent rates of both species were considerably faster than sink rates of either pelagic (unweighted = ~0.2 $m.s^{-1}$; weighted = ~0.5 $m.s^{-1}$; Melvin et al. 2013; Chapter 6) or demersal longline sink rates (unweighted = ~0.2 $m.s^{-1}$; integrated weight = ~0.5 $m.s^{-1}$; Robertson et al. 2003). During my study c. 15% of dives from both species were > 5 m, which is the current recommended depth to which pelagic longlines should be protected by bird-scaring lines (Melvin et al. 2014). Despite only c. 2% of white-chinned and grey petrel dives reaching depths > 10 m, four of the nine tagged white-chinned petrels and three of the seven tagged grey petrels made dives deeper than 10 m, suggesting many individuals from both species have the ability to dive deep. Many other seabird species regularly recorded as longline bycatch frequently dive to depths > 5 m, with many capable of reaching depths > 10 m.

As white-chinned petrels are the most commonly recorded bycatch species in Southern Hemisphere pelagic longline fisheries, and have the ability to facilitate the bycatch of other species, it is important to take their diving abilities into account when considering mitigation measure recommendations. From the results of my study, along with other seabird diving studies, I suggest that hooks should be protected to a depth of 10 m, which is deeper than recommended by Melvin et al. (2014). To protect hooks to this recommended depth, the design and implementation of seabird bycatch mitigation measures needs to be further investigated. Redesigning bird-scaring lines or reducing vessel setting speeds could ensure many more hooks are protected by the bird-scaring lines. However branchline weighting is widely recognised as the most important method of reducing seabird bycatch from pelagic longliners (Robertson et al. 2006). Despite this there is still a lack of knowledge as to how branchline weighting affects fishing operations (addressed in Chapter 6). My results help set recommendations as to what depth mitigation measures should protect baited hooks, thus improving the effectiveness and design of current and future seabird bycatch mitigation measures for longline fisheries.

Appendix 5.1a. Comparison of generalised linear mixed models (GLMM) explaining dive depths and dive durations of white-chinned petrels using sex, breeding stage and time of day/lunar luminance as explanatory variables and individual bird as a random effect.

Model	df	Loglikelihood	AICc	ΔAICc	Weight
All dives (to test time of day)					
Dive depth	-	-	-	-	-
Breeding stage	**4**	**-663.919**	**1336.0**	**0.00**	**0.237**
Breeding stage + Sex	5	-663.148	1336.5	0.53	0.182
Time of day + breeding stage	5	-663.246	1336.7	0.72	0.165
Sex	4	-664.499	1337.1	1.16	0.133
Time of day + breeding stage + sex	6	-662.609	1337.5	1.53	0.110
Null	3	-666.142	1338.4	2.39	0.072
Time of day + Sex	5	-664.206	1338.6	2.64	0.063
Time of day	4	-665.757	1339.7	3.68	0.038
Dive duration	-	-	-	-	-
Breeding stage + Sex	**5**	**-816.014**	**1642.2**	**0.00**	**0.300**
Time of day + breeding stage + sex	6	-815.399	1643.1	0.85	0.196
Sex	4	-818.191	1644.5	2.28	0.096
Breeding stage	4	-818.216	1644.6	2.33	0.093
Null	3	-819.297	1644.7	2.44	0.089
Time of day	4	-818.406	1644.9	2.71	0.077
Time of day + Sex	5	-817.395	1645.0	2.76	0.075
Night dives only (lunar luminance)					
Dive depth	-	-	-	-	-
Breeding stage	**4**	**-80.472**	**169.9**	**0.00**	**0.473**
Breeding stage + sex	5	-79.693	170.8	0.95	0.294
Sex	4	-81.954	172.9	2.97	0.107
Null	3	-83.282	173.1	3.23	0.094
Lunar luminance	4	-84.260	177.5	7.58	0.011
Breeding stage + lunar luminance	5	-83.176	177.8	7.92	0.009
Breeding stage + Lunar luminance + sex	6	-82.178	178.5	8.56	0.007
Lunar luminance + sex	5	-83.608	178.7	8.78	0.006
Dive duration	-	-	-	-	-
Breeding stage + sex	5	-113.809	239.1	0.00	0.281
Sex	**4**	**-115.217**	**239.4**	**0.30**	**0.241**
Breeding stage	4	-115.271	239.5	0.41	0.228
Null	3	-116.509	239.6	0.50	0.219
Breeding stage + lunar luminance + sex	6	-115.571	245.2	6.16	0.013
Breeding stage + lunar luminance	5	-117.249	246.0	6.88	0.009
Lunar luminance + sex	5	-117.900	247.3	8.18	0.005
Lunar luminance	4	-119.394	247.7	8.66	0.004

Appendix 5.1b. Comparison of generalized linear mixed models (GLMMs) to explain dive depth and dive duration. 'Time of day + sex' were used as variables for all dives, and 'moon + sex' as variables for night dives only. The selected models are in bold.

Model (model no.)	df	Loglikelihood	AICc	ΔAICc	Weight
All dives					
Dive depth	-	-	-	-	-
Time of day (1)	**4**	**-1227.44**	**2462.93**	**0.00**	**0.66**
Time of day + sex (2)	5	-1227.10	2464.29	1.36	0.34
Null (3)	3	-1235.41	2476.86	13.93	0.00
Sex (4)	4	-1235.21	2478.48	15.54	0.00
Dive duration	-	-	-	-	-
Time of day + sex (5)	5	-1866.89	3743.87	0.00	0.58
Time of day (6)	**4**	**-1868.44**	**3744.93**	**1.06**	**0.34**
Sex (7)	4	-1870.42	3748.90	5.02	0.05
Null (8)	3	-1871.80	3749.63	5.76	0.03
Night dives only					
Dive depth	-	-	-	-	-
Null (9)	3	-504.34	1014.77	0.00	0.29
Moon present (10)	4	-503.40	1014.94	0.17	0.26
Sex (11)	4	-503.52	1015.17	0.41	0.24
Moon present + sex (12)	5	-502.60	1015.40	0.63	0.21
Dive duration	-	-	-	-	-
Sex (13)	**4**	**-835.09**	**1678.32**	**0.00**	**0.50**
Moon present + sex (14)	5	-834.75	1679.69	1.37	0.25
Null (15)	3	-837.26	1680.60	2.28	0.16
Moon present (16)	4	-836.86	1681.85	3.54	0.09

Appendix 5.2a. The effects of the different variables on both dive depth and duration of white-chinned petrels; separate models to assess time of day (based on all dive data) and lunar luminance (restricted to nocturnal dives).

Variable	**Estimate**	**Standard error**	**z-value**	***p***
Dive depth (time of day)	-	-	-	-
Intercept	**3.316**	**0.646**	**5.118**	**< 0.001**
Breeding stage (incubation)	-1.079	0.622	1.421	0.155
Sex (male)	0.633	0.712	0.739	0.460
Time of day (night)	-0.387	0.301	1.280	0.201
Dive duration (time of day)	-	-	-	-
Intercept	**3.983**	**1.095**	**3.630**	**< 0.001**
Breeding stage (Incubation)	1.356	1.083	1.038	0.299
Sex (male)	1.376	1.082	1.055	0.292
Time of day (night)	-0.503	0.506	0.990	0.322
Dive depth (lunar luminance)	-	-	-	-
Intercept	**3.355**	**0.784**	**4.184**	**< 0.001**
Breeding stage (incubation)	-1.358	0.792	1.360	0.174
Sex (male)	0.4166	1.145	0.286	0.775
Dive duration (lunar luminance)	-	-	-	-
Intercept	**3.981**	**1.192**	**3.239**	**0.001**
Breeding stage (Incubation)	0.241	1.682	0.111	0.981
Sex (male)	-0.326	1.721	0.146	0.884

Appendix 5.2b. The effects of the different variables on both dive depth and duration of grey petrels; separate models to assess time of day (based on all dive data) and lunar luminance (restricted to nocturnal dives). Significant variables are highlighted in bold.

Variable	Estimate	Standard error	z-value	*p*
All dives				
Dive depth	-	-	-	-
Intercept	**1.162**	**0.122**	**9.500**	**< 0.001**
Time of day (Night)	**-0.172**	**0.043**	**-4.038**	**< 0.001**
Dive duration	-	-	-	-
Intercept	2.330	0.166	**14.009**	**< 0.001**
Sex (Male)	-0.410	0.241	-1.702	0.089
Time of day (Night)	**-0.130**	**0.049**	**-2.673**	**0.008**
Night dives only				
Dive depth	-	-	-	-
Intercept	**0.947**	**0.174**	**5.440**	**< 0.001**
Dive duration	-	-	-	-
Intercept	**2.281**	**0.173**	**13.196**	**< 0.001**
Sex (Male)	**-0.551**	**0.269**	**-2.049**	**0.041**

CHAPTER 6

SLIDING LEADS: A NEW, SAFE, LINE WEIGHTING TECHNIQUE TO REDUCE SEABIRD BYCATCH FOR PELAGIC LONGLINE FISHERIES

Abstract

Seabird bycatch from pelagic longline fisheries can be reduced when two out of three best practice mitigation measures are used in combination. Widespread adoption of best practice has yet to occur, perhaps because measures should be effective in reducing seabird bycatch without compromising fish catches, operational efficiency or crew safety. Unlike conventional weighted swivels, sliding leads are designed to improve safety of line weighting by sliding if the line breaks under tension. I tested 45 g and 60 g sliding leads in either black or glowing colour against unweighted (control) branchlines onboard Korean pelagic longline vessels (n = 217 000 experimental hooks), with weights placed 5 cm, 60 cm, 100 cm and 200 cm from hooks. I found no significant impact on catch rates of southern bluefin tuna when using either 45 g (p = 0.287) or 60 g (p = 0.332) glowing sliding leads placed 5 cm from the hook. The results were more complex for experimental lines set at lower latitudes when fishing for tropical and temperate tunas. Here I found no catch rate impacts for yellowfin or bigeye tunas when 45 g black and 60 g glowing leads were placed at 100 cm (p = 0.100 and 0.135, respectively), or when using 60 g glowing leads at 200 cm (p = 0.899). However tuna catch rates were reduced significantly when weights were placed at 5 cm and at 60 cm. There was no difference in flyback occurrence between treatments. Fishing operations were generally unaffected, but branchline entanglements increased significantly with 60 g at 100 cm (p = 0.037) and 200 cm (p < 0.001) from the hook; only the latter was considered problematic for the crew. Too few birds were caught in this study (n = 2) to evaluate the impact of sliding leads on seabird bycatch, however previous trails have concluded that line weighting decreases seabird bycatch. Sliding leads allow Korean-style pelagic longliners to operate safely and efficiently in compliance with regulations by using line weighting and bird-scaring lines as their seabird bycatch mitigation options.

6.1 Introduction

With the introduction of a number of mitigation measures, seabird bycatch from global fisheries has seen a significant decrease in recent years. Seabird bycatch however, still remains a problem in many fisheries worldwide, particularly the pelagic longline fishery (review by Anderson et al. 2011). Some of the most important mitigation measures which have proven to reduce seabird bycatch include: the use of bird-scaring lines (Melvin et al. 2013; Yokota et al. 2011), night-time setting (Petersen et al. 2008) and the addition of weight to branchlines (Robertson et al. 2010). Other mitigation devices such as hook pods (Sullivan 2011) and Smart Tuna Hooks (GB Baker unpubl. data)

are also effective in reducing seabird bycatch, however their practicality for some fleets requires further investigation. These measures all ensure that baited hooks are either entirely unavailable to scavenging seabirds or are available for less time during the setting process. For any mitigation device or measure to be accepted by fishing companies worldwide, crew safety, target fish catches and operational efficiency cannot be compromised (Gilman 2011; Sullivan et al. 2012). It is important to note that different fleets use very different fishing gear and techniques, therefore a seabird bycatch mitigation measure which works in one fishery may not necessarily work identically in another fleet.

Most seabird bycatch mitigation measures that have been adopted by pelagic longline fisheries can, at times, compromise fishing efficiency. This can occur as target fish are deterred by the addition of mitigation devices close to or at the hook, often for unknown reasons. Fishing efficiency can be reduced when fishing operations, either during setting or hauling, are slowed down by the application of mitigation measures/devices. Bird scaring-lines are widely used as a preferred option. Despite this, some fishing masters complain of line entanglements with bird-scaring lines, particularly when setting lines during strong cross-winds (pers. obs.). Increasing the sink rate of baited hooks, by adding weight to branchlines, is widely accepted as the most important method of reducing seabird bycatch (Robertson et al. 2006; Jiménez et al. 2013; Melvin et al. 2013, 14; ACAP 2016), yet can compromise crew safety (Sullivan et al. 2012). If a bite-off occurs during line hauling, the recoiling of the monofilament line can cause the weight to shoot back dangerously towards crew members on the hauling deck (Sullivan et al. 2012).

Sliding leads enable additional weight to be added to branchlines without additionally compromising crew safety, as the sliding leads are able to slip off branchlines in the event of a bite-off, or at least reduce the momentum of the flyback (McCormack and Rawlinson 2016). Once sliding leads have been applied to branchlines they can be easily repositioned along the branchline, to the desired distance from the hook, without having to take apart and reassemble fishing gear, as would be done when repositioning weighted swivels. Sliding leads can be encased in a luminescent nylon sheath which effectively replaces the need for glow sticks to attract fish, which are a significant source of marine pollution (Sullivan 2011).

Sliding leads were used during production fishing onboard Korean tuna longliners to test their effectiveness and practicality in reducing seabird bycatch. I had four objectives: (i) determine differences in seabird bycatch and target fish catch rates between branchlines weighted with sliding leads at differing distances from the hook and unweighted branchlines, (ii) assess crew safety implications of weighting branchlines with sliding leads, (iii) measure line sink rates of weighted and

unweighted branchlines, (iv) determine whether the use of sliding leads will have any impact on operational efficiency (e.g. increased line entanglements).

6.2 Methods

6.2.1 Fishing trips, vessels and gear setup

Research trials were conducted onboard three Korean pelagic longline vessels operating off western Australia (Vessel A and B) and southern Africa between 2014 and 2015 (Vessel C; Fig. 6.1). Fishing gear consisted of braided monofilament mainline, 40 cm diametre floats (at ~500 m intervals along mainline), radio beacons and branchlines. Branchlines were attached to the mainline at ~40 m intervals and measured ~40 m in length consisting of varying lengths of braided nylon, braided monofilament and monofilament sections, with all branchlines ending in a minimum of 5 m of monofilament with a steel leader (~40 cm) attached to the hook. No light sticks were used on any of the experimental or non-experimental sections. The number of hooks deployed between floats, is referred to as a basket; they were consistent within a trip but differed between vessels (either 11 or 12 hooks). Setting usually lasted ~5 h and generally commenced between 04h30 and 07h30, with hauling commencing ~4 h after completion of setting and generally lasting ~12 h. Roughly 3000 hooks were set daily with a target fishing depth of approximately ~150 m. Depending on target species and area, bait consisted of sardine (*Sardinops* sp.), horse mackerel (*Trachurus* sp.), Argentine squid (*Illex argentinus*) and round scad (*Decapterus maruadsi*). Vessels A and B targeted southern bluefin tuna (referred to SBT-directed vessels) off western Australia, however retained butterfly kingfish (*Gasterochisma melampus*) and blue sharks. Vessel C fished in warmer waters of the east coast of southern Africa and therefore primarily targeted tropical and temperate tunas including yellowfin, bigeye and albacore tunas (referred to as TTT-directed vessel), however also retained blue sharks, mako sharks, swordfish and marlin species (*Makaira* spp.).

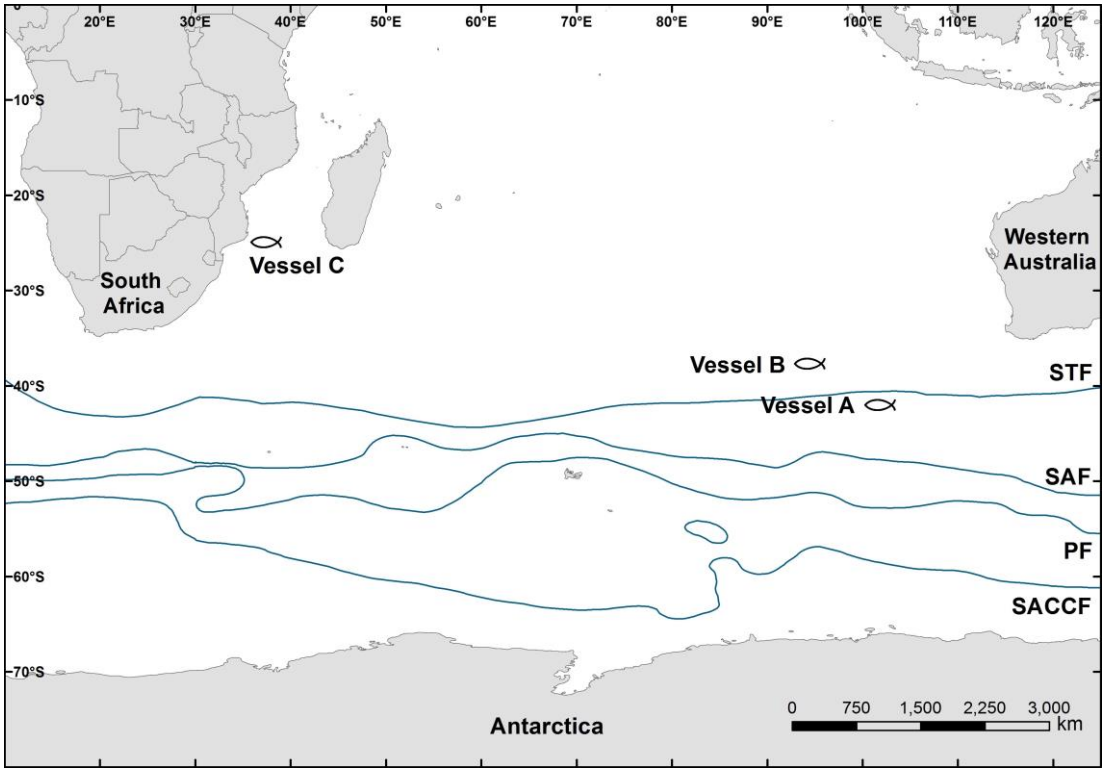


Fig. 6.1. Approximate fishing locations of the tropical and temperate tuna-directed vessel (Vessel C) and southern bluefin tuna-directed vessels (Vessels A and B), with the Subtropical Front (STF), Sub-Antarctic Front (SAF), Polar Front (PF) and Southern Antarctic Circumpolar Current Front (SACCF), all indicated.

6.2.2 Sliding leads

Sliding leads produced by Fishtek Pty Ltd., UK (www.fishtek.co.uk) and known as Lumo Leads were used throughout my study. Monofilament line of the branchlines was passed through the sliding lead and could be fixed at any distance from the hook by simply tightening the tapered screw-cap. If a branchline is put under extreme pressure, whilst a fish is being hauled, the monofilament line will stretch (10–20% before breaking) and constrict slightly; the reduced diameter of the stretched monofilament means that a snapped line will pass through the sliding lead, either resulting in the sliding lead falling off the line or dampening the energy of the recoiling line, instead of shooting back towards the vessel. Several versions of sliding leads were tested, varying in mass and nylon coating colour: a black coating (hereafter known as black sliding leads) of mass 45 g and a luminescent nylon coating (hereafter known as glowing sliding leads) of masses 45 g and 60 g. Unfortunately I was unable to secure black 60 g sliding leads for my sea trials and thus these sliding leads were not tested in my study.

6.2.3 Experimental design

Treatments (weighted or unweighted) were arranged in alternating baskets which ensured that treatment sizes remained consistent. Trials onboard SBT-directed vessels only used two-way experiments between weighted and unweighted baskets (Fig. 6.2a) while trials onboard Vessel C used both two- and three-way experiments (Fig. 6.2b). It is important to note that a basket never included a mix of treatments, only one treatment per basket. For SBT-directed sets (Vessels A and B), glowing sliding leads (45 g and 60 g) were tested at a distance of 5 cm from the hook only. For Vessel C, 60 g glowing and 45 g black sliding leads were tested at varying distances from the hook (5 cm, 60 cm, 100 cm and 200 cm; Table 6.1). For all sets on which sliding leads were positioned 5 cm from the hook (Vessels A, B and C), sliding leads were attached to the wire tracers which meant that the sliding leads were unable to slide or fall off the line during a bite-off. Despite this the sliding leads were applied to the wire tracers as it was important to understand how weight close to the hook affected fish catches. For those sets testing sliding leads > 5 cm from the hook, sliding leads were attached to the monofilament line above the wire trace.

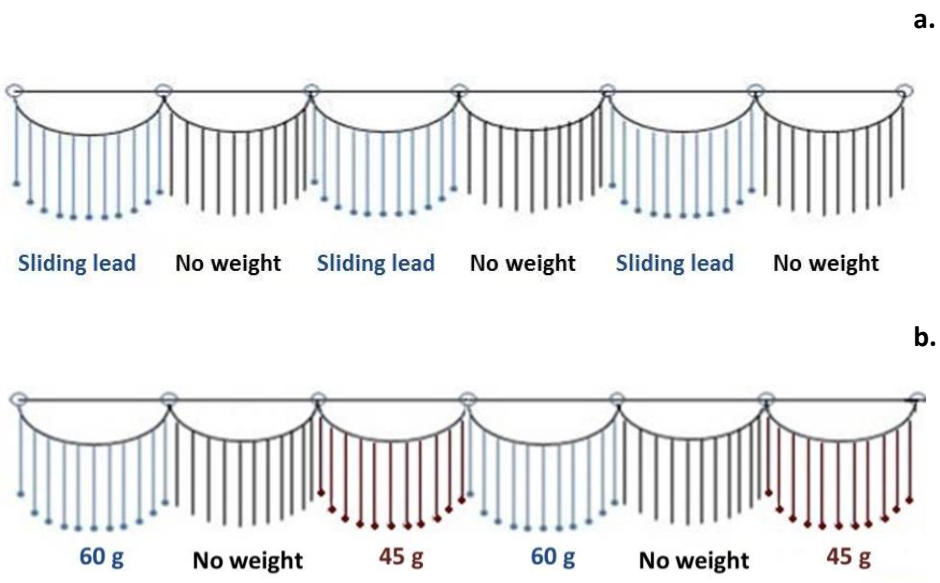


Fig. 6. 2. Sliding lead configuration of a. two-way trials between weighted and unweighted baskets (as performed by all vessels), and b. three-way trials between two different sliding leads and unweighted baskets (as performed by Vessel C only).

Table 6.1. Experimental configuration and trip information for the four vessels used to conduct sliding lead trials between 2014–2015, targeting southern bluefin (SBT), yellowfin (YFT), bigeye (BET) and albacore tunas (ALB). * Sets testing a combination of black and glowing sliding leads against unweighted control baskets; sliding leads were never mixed within baskets.

Vessel	A	B	C	
Sliding lead mass (g)/colour	45/glowing	60/glowing	45/black	60/glowing
Distance from hook (cm)	5	5	5, 60, 100	100, 200
Experimental sets/baskets	48/2410	59/6250	23/3540 10/1445 45/4477*	30/2996* 23/2375*
Fishing areas	37 – 39° S 100 – 103° E	30 – 38° S 83 – 103° E	19 – 29° S 36 – 37° E	
Target species	SBT	SBT	ALB, YFT, BET	

6.2.4 Setting and hauling data

Data including GPS co-ordinates, time, wind speed and swell height were collected from vessel logbooks for the start and end of each set and haul. During the haul observers positioned themselves on the haul deck with a clear view of the hooks as they were brought onboard. All experimental branchlines were observed and for hooks which caught fish, the following were recorded; treatment (weighted or unweighted), basket number, species, mass, condition of catch (scavenged/intact), catch fate (retained/discarded) and processing method (if retained). Observers onboard SBT-directed vessels recorded both retained and discarded fish catch per set, while observers onboard TTT-directed vessels recorded retained and discarded fish catch at a finer scale (per basket) for all sets.

6.2.5 Hook sink rate

Sink rate of weighted and unweighted treatments were measured using G5 Cefas TDRs (35 x 11 mm; 6.2 g) and G5 Host software, onboard Vessel C only. The TDRs were taped to the monofilament branchline using waterproof Tesa tape ~40 cm from the hook (~10 cm above the wire trace) and were programmed to sample depth and temperature every 1 s. Water entry time was accurately recorded using a digital wristwatch and the number of seconds to reach a depth of 10 m was obtained from the data file.

6.2.6 Impact on operational efficiency

Fishing masters have suggested sliding leads are likely to have negative effects on fishing operations by increasing bait loss and branchline entanglements during setting operations. Therefore both of these factors were assessed on Vessel C by recording rates of bait loss during line setting and branchline entanglements upon line hauling. During line setting, observers monitored a portion of the experimental section (for both weighted and unweighted treatments) from above the setting deck and recorded bait loss upon branchline entry into water after line casting. If, upon entry into water, bait had become dislodged from the hook it would remain floating on the surface instead of rapidly sinking and thus could be easily observed.

To determine branchline entanglements, observers monitored sections of both weighted and unweighted treatments during line hauling and all branchline entanglements were noted (Vessel C only). A branchline was considered 'entangled' if upon line hauling any section of the branchline contained a knot or was tangled in any way. Any branchlines identified as entangled by the hauling crew were put aside to be untangled by other members of the crew and hence the entangled branchlines could be easily identified.

During hauling operations some sliding leads were noted to have moved up or down the monofilament line (Vessel C only). For branchlines where the sliding lead had shifted significantly, crew members were asked to reposition the sliding lead to the correct distance from the hook whilst hauling and coiling branchlines. On completion of sliding lead trials (100 cm or 200 cm from the hook), 200 weighted branchlines from each treatment were put aside to quantify sliding lead movements relative to the hook.

6.2.7 Crew safety

To test the safety aspect of sliding leads, several parameters for all flyback events were recorded. A flyback was defined as a line break or accidental loss of fish during line hauling which caused the branchline to shoot back towards the vessel. Data collected included; treatment, fish species hooked, whether line was intentionally or unintentionally broken, where along the line the break occurred, where the hook/weight landed/struck, whether the sliding lead slipped off the line and if any crew were injured.

6.2.8 Statistical analyses

All statistical analyses were conducted using the R statistical package (R Core Team 2015). T-tests were performed to test for differences in bait loss rate between different weighted treatments and unweighted branchlines. T-tests were also performed to test for differences in entanglement rates between the different weighted treatments and unweighted branchlines. A non-parametric Kruskal-Wallis test was used to compare line sink rates between the different treatments at three depth strata; 0–2 m, 2–4 m, 4–10 m and 0–10 m. A Dunn test was used to determine differences between the individual groups.

To understand the impact of different line weighting treatments on fish catches, SBT- and TTT-directed vessels were analysed separately. As fish catch data were recorded at a different scale (catch per set versus catch per basket), this further enforced the decision to treat these data separately. Generalised linear models (GLMs) with a Poisson distribution (logarithmic link) were developed for SBT-directed vessels (Sullivan et al. 2012), while GLMS with a negative-binomial distribution (logarithmic link) were used for analysing catches from the TTT-directed vessel (Hamel et al. 2009). Target catch was the response variable for all models; for SBT-directed vessels target catch included southern bluefin tuna only, while for the TTT-directed vessels, albacore tuna and the combined catch of yellowfin and bigeye tuna were modelled separately. A full suite of models, with all variables and two-way interaction terms were constructed, and the model with the lowest AIC score was selected as the most appropriate model. As two SBT-directed vessels were used in these trials, vessel identification was included as a variable for SBT-directed models, however vessel identification was excluded for TTT-directed models (one vessel only).

6.3 Results

6.3.1 Impact on catch and bycatch

Experimental effort totalled 13 832 experimental baskets (or > 150 000 experimental hooks) using glowing sliding leads onboard SBT-directed vessels (Table 6.1). The use of glowing sliding leads (both 45 and 60 g) 5 cm from the hook did not significantly reduce target catch of SBT-directed vessels; target catch was only influenced by vessel identification (Table 6.2). Overall southern bluefin tuna catch rates when using 45 g sliding leads 5 cm from the hook (6.8 fish per 1000 hooks) were similar to unweighted branchlines (5.8 fish per 1000 hooks; Fig. 6.3), as were catch rates when using 60 g

sliding leads 5 cm from the hook (weighted = 5.1 fish per 1000 hooks; unweighted = 6.1 fish per 1000 hooks; Fig. 6.3).

Table 6.2. The effect of different variables for the best selected models. Significant variables are highlighted in bold. * The best selected model did not include Treatment as a variable for bluefin tuna-directed vessels.

Variable	Estimate	SE	z-value	*p*
*Southern bluefin tuna**	-	-	-	-
Intercept	**-5.181**	**0.082**	**-63.246**	**> 0.001**
Vessel ID: B	**-0.580**	**0.111**	**-5.232**	**> 0.001**
Yellowfin & bigeye tuna	-	-	-	-
Intercept	**-3.626**	**0.159**	**-22.742**	**> 0.001**
Experiment: Sliding lead; 60 cm	**1.862**	**0.208**	**8.949**	**> 0.001**
Experiment: Sliding lead; 100/200 cm	**1.375**	**0.197**	**6.986**	**> 0.001**
Experiment: Sliding lead; 100 cm	**1.475**	**0.185**	**7.970**	**> 0.001**
Experiment: Sliding lead; 200 cm	**1.215**	**0.229**	**5.305**	**> 0.001**
Treatment: 45 g/5 cm/black	-0.323	0.243	-1.329	0.184
Treatment: 45 g/60 cm/black	**-0.528**	**0.203**	**-2.596**	**0.009**
Treatment: 45 g/100 cm/black	-0.201	0.121	-1.649	0.099
Treatment: 60 g/100 cm/glowing	-0.215	0.144	-1.494	0.135
Treatment: 60 g/200 cm/glowing	-0.021	0.161	-0.127	0.899
Albacore tuna	-	-	-	-
Intercept	**-1.099**	**0.046**	**-24.035**	**> 0.001**
Experiment: Sliding lead; 60 cm	**-2.067**	**0.188**	**-11.002**	**> 0.001**
Experiment: Sliding lead; 100 cm	**-3.188**	**0.207**	**-15.424**	**> 0.001**
Experiment: Sliding lead; 100/200 cm	**-3.160**	**0.240**	**-13.159**	**> 0.001**
Experiment: Sliding lead; 200 cm	**-2.322**	**0.244**	**-9.518**	**> 0.001**
Treatment: 45 g/5 cm/black	**-0.866**	**0.081**	**-10.718**	**> 0.001**
Treatment: 45 g/60 cm/black	**-0.657**	**0.311**	**-2.114**	**0.035**
Treatment: 45 g/100 cm/black	-0.185	0.261	-0.707	0.479
Treatment: 60 g/100 cm/glowing	-0.326	0.328	-0.992	0.321
Treatment: 60 g/200 cm/glowing	-0.458	0.313	-1.463	0.143

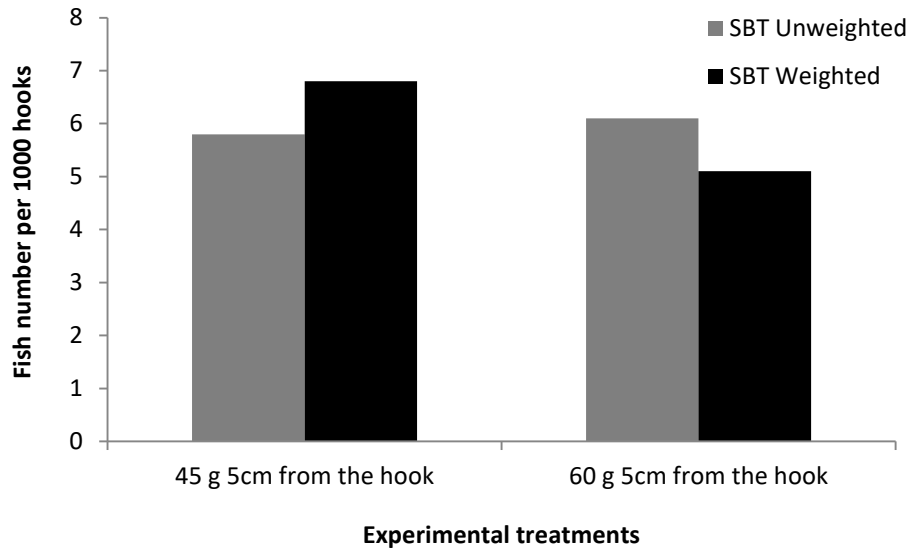


Fig. 6.3. Summary of southern bluefin tuna catch between different treatments for southern bluefin tuna-directed vessels (Vessels A and B).

Trials onboard Vessel C targeting temperate and tropical tunas amounted to 13 335 experimental baskets (or > 150 000 experimental hooks). Only two experimental treatments (45 g black sliding leads 5 cm and 60 cm from hook) significantly decreased target catch rates (Table 6.2 and 6.3). The combined catch of yellowfin and bigeye tuna was reduced significantly when using 45 g black sliding leads 60 cm from the hook (15.8 vs 9.3 fish per 1000 hooks; $p = 0.009$; Fig. 6.4a) and albacore tuna was significantly reduced when using 45 g black sliding leads at 5 both cm (11.6 vs 27.7 fish per 1000 hooks; $p < 0.001$) and 60 cm from the hook (3.9 vs 2.0 fish per 1000 hooks; $p = 0.035$; Fig. 6.4b). Target catch rates (bigeye/yellowfin tuna and albacore tuna) on weighted branchlines were not significantly different when using sliding leads (both 45 and 60 g) 100 cm from the hook. Target fish catch rates were statistically indistinguishable between weighted and unweighted branchlines when using 60 g glowing sliding leads 200 cm from the hook. Fish bycatch (defined as any catch that was not retained) was significantly reduced when using 45 g black sliding leads at both 5 cm ($p = 0.014$) and 60 cm from the hook ($p = 0.005$), while no significant differences were noted for any other treatments.

Table 6.3. Summary of target fish catch numbers and rates for vessels targeting southern bluefin tuna (SBT; Vessels A and B) and tropical and temperate tunas; albacore (ALB), yellowfin (YFT) and bigeye tuna (BET; Vessel C).

Vessel	Treatment (mass/cm from hook/colour)	Hooks	Species	Control total (fish/1000 hooks)	Black total (fish/1000 hooks)	White total (fish/1000 hooks)
A	45 g/5/glowing	26 510	SBT	81 (6.1)	-	68 (5.1)
B	60 g/5/glowing	28 446	SBT	83 (5.8)	-	96 (6.8)
C	45 g/5/black	42 480	YFT & BET	47 (2.2)	34 (1.6)	-
C	45 g/5/black	42 480	ALB	588 (27.7)	247 (11.6)	-
C	45 g/60/black	15 930	YFT & BET	126 (15.8)	74 (9.3)	-
C	45 g/60/black	15 930	ALB	31 (3.9)	16 (2.0)	-
C	45 g/100/black	49 428	YFT & BET	253 (10.3)	208 (8.4)	-
C	45 g/100/black	49 428	ALB	31 (1.3)	26 (1.1)	-
C	60 g/100/glowing	32 956	YFT & BET	179 (10.9)	-	142 (8.6)
C	60 g/100/glowing	32 956	ALB	23 (1.4)	-	16 (0.9)
C	60 g/200/glowing	26 126	YFT & BET	116 (8.9)	-	117 (9.0)
C	60 g/200/glowing	26 126	ALB	23 (1.8)	-	16 (1.2)

Only two birds were caught during the SBT-directed trials, with no birds caught during TTT-directed trials. Both were caught while testing 60 g glowing sliding leads 5 cm from the hook (1 unweighted branchline; grey-headed albatross, 1 weighted branchline; black-browed albatross).

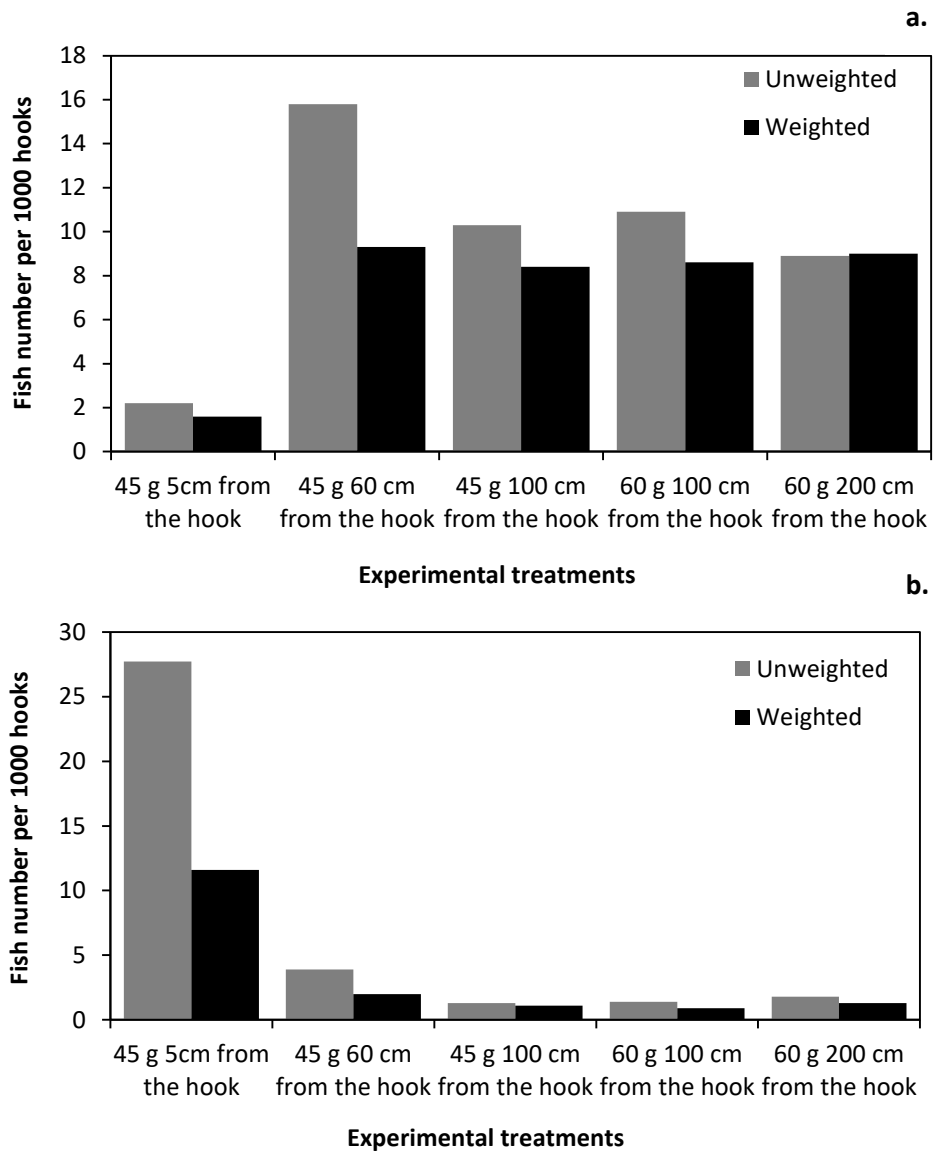


Fig. 6.4. Summary of fish catches between different treatments of the tropical and temperate tuna-directed vessel for a. combined yellowfin and bigeye tuna and b. albacore tuna.

6.3.2 Hook sink rate

Useable TDR data were retrieved from 111 TDR deployments of six different treatments (Table 6.4). Differences in sink rates at different depth strata were not apparent from my study and thus sink rate to 10 m only was recorded. Significant differences in sink rate to 10 m were found between unweighted hooks and all weighted treatments (Table 6.4), however no significant differences were found between sink rates of any of the weighted treatments. Unweighted branchlines took an additional 10 s to reach a depth of 10 m compared to the slowest weighted treatment (60 g 200 cm from the hook).

Table 6.4. Summary of TDR sink rate results to a depth of 10 m, from Vessel C

Treatment	*n*	Mean sink rate ($m.s^{-1}$)	Seconds to 10 m	*p* (Unweighted vs weighted)
Control (unweighted)	41	0.27	37.0	-
45 g at 5 cm from hook	15	0.41	24.4	< 0.001
45 g at 60 cm from hook	14	0.39	25.6	< 0.001
45 g at 100 cm from hook	9	0.40	25.0	< 0.001
60 g at 100 cm from hook	11	0.38	26.3	< 0.001
60 g at 200 cm from hook	21	0.37	27.0	< 0.001

6.3.3 Impact on operational efficiency

Bait loss was observed during the setting of 14 751 hooks. No significant differences were found between any of the weighted treatments (45 g 100 cm from hook, 60 g 100 and 200 cm from hook) and unweighted (control) treatments, with very low levels of bait loss during setting (c. 1%) for all treatments (Table 6.5).

Table 6.5. Summary of bait loss between different weighting treatments, with the significant difference (*p*) between unweighted branchlines indicated.

Treatment	Hooks observed	Bait loss (%)	*p*
Control (unweighted)	4048	51 (1.3)	-
45 g at 100 cm from hook	4048	49 (1.2)	0.884
60 g at 100 cm from hook	1441	14 (1.0)	0.223
60 g at 200 cm from hook	2607	31 (1.2)	0.856

In total 26 216 branchlines were closely monitored during line hauling to check for entanglements from four treatments onboard Vessel C only: unweighted (control), 45 g 100 cm from hook, 60 g 100 cm from hook and 60 g 200 cm from hook (Table 6.6). Entanglement rates did not differ statistically between branchlines weighted with 45 g 100 cm from the hook (9.0%) and unweighted branchlines (7.5%; $p = 0.090$), however did differ significantly between branchlines weighted with 60 g 100 cm from the hook (10.8 %; $p = 0.036$) and 60 g 200 cm from the hook (21.0%; $p < 0.001$), compared to unweighted branchlines. Very few of the sliding leads had shifted significant distances (> 25 cm) from the original positions for either 45 g 100 cm (13%) or 60 g 200 cm (14%).

Table 6.6 Summary of entanglements between different weighting treatments, with the significant difference (*P*) between unweighted branchlines indicated.

Treatment	Hooks observed	Entanglements (%)	*p*
Control (unweighted)	16 261	1217 (8 %)	-
45 g at 100 cm from hook	10 175	920 (9 %)	0.090
60 g at 100 cm from hook	4589	497 (11 %)	0.036
60 g at 200 cm from hook	6086	1277 (21 %)	< 0.001

6.3.4 Crew Safety

A total of 17 flybacks (14 line breaks; 3 accidentally unhooked fish) were observed during hauling (onboard Vessel C) which could have compromised crew safety. These potentially dangerous line breaks occurred on both weighted (40% black sliding leads; 30% glowing sliding leads) and unweighted branchlines (30%), with all but one involving hooked sharks. Most (82%) of the flybacks occurred when the fish/shark was right alongside the hauling station and the line was purposely broken to avoid hauling the shark aboard before dehooking. One minor injury resulted from a flyback of an unweighted branchline, which was purposely broken. All three of the accidentally unhooked fish were on weighted branchlines, however none of these resulted in dangerous flybacks; two of the branchlines lost all momentum in the water before reaching the vessel while one branchline flew back onto the hauling deck however had lost nearly all momentum on landing.

6.4 Discussion

6.4.1 Impact on catch and bycatch

The results of the branchline weighting trials onboard SBT-directed vessels were encouraging: the use of sliding leads did not affect target fish catch rates. However the effect of branchline weighting on TTT-directed catch was more complex. There was a significant reduction in albacore tuna catch when using 45 g black sliding leads close to the hook (5 cm or 60 cm from the hook), with no other sliding lead treatments having any significant effect on albacore tuna catch. Yellowfin and bigeye tuna catch was only significantly reduced when using 45 g black sliding leads 60 cm from the hook, no other sliding lead treatment significantly affected target catches, however there was a slight reduction when using 45 and 60 g sliding leads 100cm from the hook. Catch rates for yellowfin and bigeye tuna were most similar when using 60 g glowing sliding leads 200 cm from the hook (weighted = 9.0 fish per 1000 hooks; unweighted = 8.9 fish per 1000 hooks).

The fishing industry is cautious of applying weight close to the hook due to concerns that this will negatively impact fish catches (Robertson et al. 2013). Robertson et al. (2013) did not find any difference in catch rates (bigeye, yellowfin or albacore tuna) between branchlines using 40 g at the hook compared to branchlines using 60 g 3.5 m from the hook. Those results are contrary to my findings, which show weight at or near the hook reduces albacore tuna catch as well as the combined catch of yellowfin and bigeye tuna. Robertson et al. (2013) used glowing sliding leads, therefore it is possible that sliding lead colour influenced fish catches; all trials conducted during my TTT-directed trials with weight close to the hook used black sliding leads only. Gianuca et al. (2013) performed weighting trials onboard TTT-directed vessels and tested catch rates between branchlines using weighted swivels 5.5 m and 2 m from the hook. They found no differences in catch rates of target species, however yellowfin tuna catch was affected, with higher catch rates on lines with weight 2m from the hook. Melvin et al. (2013) did not find any difference in target catch rate between weighted (60 g safe leads 70 cm from the hook) and unweighted branchlines while onboard a TTT-directed vessel.

No birds were caught on the TTT-directed vessel, likely because almost all fishing was conducted in the warmer waters of the low latitudes with very low seabird abundance. Previous research suggests that due to increased line sink rates, sliding leads should reduce seabird bycatch rates; studies conducted in waters with very high seabird abundance and attendance Jiménez concluded that seabird bycatch rates are much higher on unweighted branchlines compared to weighted branchlines (Melvin et al 2013; Jiménez et al 2013). Attacks on baited hooks were reduced by as much as 59% when using weight 1 m from the hook compared to weight 4.5 m from the hook (Jiménez et al. 2013). Due to the low numbers of seabirds killed during this study, I cannot assess the efficacy of sliding leads at reducing seabird bycatch. Despite SBT-directed vessels fishing in areas where high seabird abundances are common, just two birds were killed from > 83 000 hooks. Korean-style unweighted branchlines sank relatively fast compared to other studies (e.g. Melvin et al. 2013; Jiménez et al. 2013), which may explain some of the low observed seabird bycatch.

6.4.2 Hook sink rate

The sink rate of baited hooks depends on both the amount of weight applied to branchlines as well as the distance of the weight to the hook (Robertson et al. 2010). Line sink rates can be divided into two phases; the initial stage when line sink rates are comparatively slow, and the final stage when line sink rates are faster (Robertson et al. 2010). Initial sink rate depends on proximity of weight to

hooks (branchlines sink faster the nearer weights are placed to the hook) while final sink rate depends on the mass of the weights (Robertson et al. 2010).

Differences in line sink rate could not be identified in my study, with sink rates to a depth of 10 m similar across all weighted treatments. All treatments of weighted branchlines sank significantly faster than unweighted branchlines, with no difference noted within the weighted treatments. Average sink rates of unweighted branchlines from my study were considerably faster (0.27 $m.s^{-1}$) than sink rates of unweighted branchlines (0.19 and 0.16 $m.s^{-1}$) from previous studies (Anderson and Mcardle 2002; Melvin et al. 2013). Melvin et al. (2013) found that line sink rate differed between branchlines weighted with 40 g and 60 g (both 1 m from the hook), thus it is perhaps surprising that line sink rates did not differ between branchlines weighted with 45 g and 60 g 1 m from the hook in my study. It is generally accepted that hooks should be protected to a depth of 10 m, a depth to which a number of bycatch-prone seabird species are able to dive (Melvin et al. 2013; Favero et al. 2016; Chapter 5). At sink rates achieved in my study, if a vessel were to set lines at a speed of 9 kn, unweighted hooks would reach a depth of 10 m at a distance of 167 m, compared to a distance of 112 and 122 m for the fastest (45 g 100 cm from the hook) and slowest sinking weighted treatments (60 g 200 cm from the hook) from my study. Thus a bird-scaring line achieving 100 m aerial coverage would respectively protect 60, 82 and 89% of hooks, when using unweighted branchlines, branchlines weighted with 60 g 200 cm from the hook and branchlines weighted with 45 g 100 cm from the hook.

6.4.3 Impact on operational efficiency

Fishing masters suspected that bait loss during line setting would be a problem when using sliding leads. They assumed that as the bait was cast, either by hand or a bait-casting machine, it would be dislodged due to the 'hinge' effect created as the weight overtakes the baited hook into the water. In my study I found no difference in bait loss during line casting between unweighted and weighted treatments (45 g and 60 g 100 cm from hook; 60 g 200 cm from hook). Due to the lack of a 'hinge effect' for treatments with weight close to the hook, bait loss is unlikely to be an issue for these treatments, however this was not tested.

Entanglements only became noticeably more frequent when sliding leads were furtherest (200 cm) from the hook; branchlines on this treatment were three times more likely to become entangled than unweighted branchlines. Although weighted branchlines (both 45 g and 60 g) 100 cm from the hook became entangled at higher rates than unweighted branchlines, the difference between the

unweighted branchlines was not significant and did not add considerable time to hauling operations (unlike 60 g 200 cm from the hook). When weight was positioned further from the hooks, the movement of the branchlines, as they were cast, was less predictable and many more sliding leads slipped through loops in the line, resulting in entanglements. During trials with sliding leads closer to the hook (5–60 cm) entanglements were rare and thus the problem of line entanglements was not mentioned by fishing masters. The crew alternated between using a bait-casting machine and hand casting when setting weighted branchlines. There were no problems with using the bait-casting machine for treatments with sliding leads 5–60 cm from the hook, however the crew preferred to hand-cast the lines when sliding leads were further away from the hook (100 and 200 cm). For treatments with sliding leads further from the hook, hand-cast lines seemed to result in a smoother cast of the line compared to machine-cast lines; the latter appeared to create a 'hinge effect' before hitting the water.

Similar to line weighting trials conducted by Melvin et al. (2013), fishing masters raised concerns during my trials that increased entanglements would reduce fish catches. Entanglements create additional work for the hauling crew, who need to disentangle or unknot lines before they can be coiled. However in this study, the vast majority of tangles consisted of a single slip-knot on the line, which was a trivial matter for the crew to repair. Crew from trials by Melvin et al. (2013) complained that weighted branchlines were cumbersome to coil, this was also a complaint by crew onboard the two SBT-directed vessels, however the crew of the TTT-directed vessel thought that the weighted branchlines represented no additional effort to coil.

As sliding leads have the ability to be shifted along the branchline to specific distances from the hook, there is the concern that sliding leads may accidentally shift position at some stage during the fishing process, which may affect fishing efficiency, or that they could be purposely shifted by crew (closer to the hook) which could negatively affect seabird bycatch . To ensure sliding leads were kept at the required distance from hooks, the crew would routinely adjust sliding leads to the correct position when coiling them after hauling (under the watch of observers); this did not complicate hauling operations and had little or no effect on hauling time. During my study only a small number of sliding leads had shifted position significantly (> 25 cm) and thus sliding lead slippage is unlikely to be a problem.

As with any commercial venture, cost is an extremely important factor. Although sliding leads such as Lumo Leads are more expensive than traditional weighted swivels, the difference is negligible. After the initial bulk order to weight the branchlines with sliding leads, further costs

would only be to ensure additional sliding leads are available onboard to reweight branchlines which have lost sliding leads.

6.4.4 Crew Safety

The addition of sliding leads to branchlines was not observed to compromise crew safety onboard Vessel C. Unfortunately the effect of sliding leads on crew safety was not recorded from Vessels A and B, however neither observers or captains noted any safety concerns of the sliding leads. Nearly all flybacks occurred when sharks were right alongside the hauling station with the crew applying tension to the branchline, often resulting in the hook being ripped out of the shark's mouth, or the breaking of the monofilament line; these actions are sometimes referred to as tear-outs. These actions would often result in the branchline shooting back towards the vessel; however these were intentional actions by the crew. During these intentional tear-outs, an unweighted branchline appeared just as likely to flyback and cause injury to the crew. On all observed accidental flybacks, branchlines had either lost all momentum before reaching the vessel, or had lost nearly all of their momentum before landing on the hauling deck, as such none of these observed accidental flybacks posed any danger to the crew. Neither the fishing master nor the crew appeared particularly concerned of the potential danger of using sliding leads throughout the trip, possibly as wire tracers were used on all branchlines, which are believed to reduce the possibility of flybacks. These results are similar to the findings of Sullivan et al. (2012) who concluded that Safe Leads (a precursor to sliding leads which used a similar 'sliding lead' concept) significantly reduced the danger of flybacks, compared to the use of weighted branchlines.

6.4.5 Future research

Despite a large number of trials conducted on different sliding lead weighting regimes there is still the need for more research on branchline weighting to better understand how they affect fish catches and seabird bycatch. Only two differently coloured sliding leads were trialled in my study, however the effect of sliding lead colour (particularly glowing versus non-glowing) on fish catches is still not understood and thus future trials to investigating sliding lead colour are recommended. Research testing glowing sliding leads close to the hook will help understand whether the colour of the black sliding leads affected fish catches onboard the TTT-directed vessel, rather than the application of weight close to the hook. To better understand the effect of sliding leads on seabird bycatch, further trials in areas of high seabird bycatch from pelagic longliners are required. Seabird

bycatch and abundance as well as seabird attack rates (during line setting) should all be investigated. As there are large differences in fishing technique, crew and other factors between different longline fleets it is important that further investigation be conducted from different fleets around the world. Fishers need to become comfortable with new fishing technology, and how it can be incorporated into their specific fishing practise, before these new technologies will be accepted by fisheries companies.

6.4.6 Recommendations

Due to low seabird bycatch from my study I was unable to assess the effect of sliding leads on seabird bycatch, however results from previous studies have proven the effectiveness of line weighting as a seabird mitigation measure (Favero et al. 2016). As a number of different sliding lead treatments did not compromise fish catches, operational efficiency or crew safety, I suggest that sliding leads be used by pelagic longline fisheries to reduce seabird bycatch. Sliding lead mass and distance from the hook are important factors which affects line sink rates, fish catches and operational efficiency. As glowing sliding leads applied < 5 cm from the hook did not affect southern bluefin tuna catch rates, fishing operations or crew safety, I recommend sliding leads be used by SBT-directed vessels.

For TTT-directed vessels fishing in areas of high seabird abundance (south of 25^{o} S) I recommend that 60 g sliding leads be placed 100 cm from the hook. Results show that this weighting regime will ensure fish catches are unaffected, while still maintaining high sink rate speeds, without an increase in bait loss. Entanglements were higher than unweighted branchlines however the difference was minor (3%) and would not significantly affect operational efficiency. These recommendations are in line with recommendations made by the Agreement for the Conservation of Albatrosses and Petrels (ACAP), which suggests the use of weight 40 g or greater within 0.5 m of the hook or 60 g or greater within 1 m of the hook (Favero et al. 2016).

CHAPTER 7

SYNTHESIS: RECOMMENDATIONS TO REDUCE SEABIRD BYCATCH FROM PELAGIC LONGLINERS OFF SOUTH AFRICA

7.1 Overview

It has been almost 30 years since seabird bycatch from pelagic longliners was recognised as a conservation concern (Brothers 1991), yet it still remains a significant source of mortality for many seabird species throughout the world's pelagic longline fleets. In other fisheries such as trawl and gillnet, and to some extent demersal longlining, seabird bycatch has been reduced to manageable levels in much shorter time periods. Seabird bycatch from pelagic longlining is a complex problem for which there is no 'silver bullet' solution but rather an array of options and measures that have to be used together to alleviate the problem. This thesis expands our knowledge of seabird bycatch from both South African and Asian longliners, and should form the basis for future seabird bycatch work off South Africa.

Studies documenting seabird bycatch depend on either researchers or observers to collect data, and because this involves spending long periods at sea, these data usually are collected by fisheries observers. As many of the world's pelagic longline fleets have only partial observer coverage (Anderson et al. 2011), it makes those studies with high levels of observer coverage more meaningful for tackling seabird bycatch. Two previous studies summarised seabird bycatch from pelagic longliners off South Africa for 1998–2005 (Ryan et al. 2002; Petersen et al. 2009b). During this period, observer coverage was low as it was not a permit condition for Asian-flagged vessels to fish with South African fisheries observers onboard. As such, their estimates of bycatch were extrapolated from observations of c. 1% (Ryan et al. 2002) and c. 10% (Petersen et al. 2009b) of total fishing effort. Following these studies, permit conditions were amended; from 2007 all foreign pelagic longline vessels fishing under South African joint venture permits were required to have a South African fishery observer onboard. As Asian vessels set roughly twice the number of hooks as South African vessels, this meant my research from Chapter 2 was based on a much high proportion of observed effort than previous studies (69% of total fishing effort; 2006–2013), thus providing the most comprehensive summary of seabird bycatch by pelagic longline vessels off South Africa. Also importantly I am able to assess change in seabird bycatch linked to changes in mitigation measures over time for the Asian fishery.

My summary of seabird bycatch in Chapter 2 forms the basis of my thesis as it identifies the magnitude of the problem, species affected and factors affecting seabird bycatch by pelagic longliners off South Africa. These topics are further addressed in Chapters 3–6, which assess how mitigation measures might be useful for reducing seabird bycatch off South Africa and globally. The conservation implications for many species have been addressed thoroughly elsewhere, thus I do not dwell

excessively on this aspect, but rather take it as read that the species-level conservation consequences are non-trivial. My results show that seabird bycatch rates off South Africa have reduced by 70% from the 0.44 birds per 1000 hooks reported by Petersen et al. (2009b). However the numbers of birds killed are still cause for concern, especially as several threatened or near-threatened species are affected. Chapter 3 further explains the composition of the bycatch, particularly the high levels of the predominant bycatch species, the white-chinned petrel (Chapters 4 and 5) as well as proving the efficacy of a new seabird bycatch mitigation device, which could be incorporated into fisheries off South Africa (Chapter 6). By understanding the scale and reasons behind pelagic longline bycatch off South Africa, more informed conservation actions can be taken.

7.2 Mitigation measures applicable to the South African pelagic longline fishery

In recent years, the permit conditions controlling pelagic longline fishing in South Africa have been amended to include a number of seabird bycatch mitigation measures. In order to further reduce seabird bycatch it is important for decision makers to regularly consult scientific literature reporting the feasibility and effectiveness of seabird bycatch mitigation measures, as well as studies describing the ecology of bycatch-vulnerable seabird species. Findings from my research are discussed below in the context of their impact on the effectiveness of seabird bycatch mitigation measures and their applicability to pelagic longline fisheries off South Africa.

7.2.1 Fisheries observers

One of the most important amendments to South African permit conditions was the mandatory deployment of fisheries observers onboard all foreign-flagged vessels fishing under South African joint venture permits in 2007. This meant that all foreign-flagged pelagic longline fishing trips were monitored; reporting fish catches, seabird bycatch and compliance with mitigation measures. Unfortunately, fisheries observers are not mandatory onboard local South African vessels, and thus observer coverage from these vessels was low (6% overall from 2006–2013), and there was no observer coverage at all from 2011–2013. This low level of observer coverage means that the magnitude of seabird bycatch by the South African fleet is poorly understood. One of the most significant findings from my research was that South African pelagic longline vessels are catching birds at unacceptably high

rates. Extrapolation to the rest of the unobserved fleet indicates that South African vessels are responsible for considerably more seabird mortality than previously estimated (Petersen et al. 2009b). It is therefore imperative that observer coverage of this fleet be increased greatly, as this will shed further light on the levels of bycatch within this fleet, and more importantly the levels of mitigation measure compliance which will affect seabird bycatch. As these vessels are generally much smaller than Asian-flagged vessels, supporting smaller fishing crews, the imposition of additional personnel, who do not contribute to fishing activities, is felt more acutely than onboard the larger, Asian vessels. For the South African fleet a potentially cheaper and more appropriate option would be the introduction of electronic monitoring. Research still needs to be conducted to determine whether this is feasible in the South African fishery, but video recording has been successfully trialed in a number of fisheries worldwide (e.g. Piasente et al. 2012). Electronic monitoring can also ensure that the entire set and haul is monitored, without the need for breaks and sleep, thus potentially ensuring more accurate estimates of seabird bycatch.

7.2.2 Vessel-specific bycatch limits

The year 2007 saw the highest levels of seabird bycatch from the South African and foreign-flagged fleets. In response, authorities introduced new seabird bycatch regulations into permit conditions in 2008, restricting vessels to a maximum mortality of 25 birds per fishing season (DEAT 2008). This measure resulted in a significant reduction in seabird bycatch in 2008, without any other meaningful change to the fishery or bycatch regulations; clearly the strong incentive to minimise seabird bycatch ensured that fishing masters and crew were either more diligent with the use of known seabird bycatch mitigation measures, or took steps to ensure that bycatch wasn't fully reported. Unfortunately the actions required of vessels which exceeded seabird bycatch limits (which were tantamount to penalties but not overtly constructed as such) were changed after 2008 to reduce the impacts on operations (and consequently reduce the 'costs' to individual vessels); this lack of enforcement apparently reduced crew vigilance in subsequent years, when seabird bycatch once again increased (but not reaching the high levels observed prior to 2008). I believe this permit regulation is an effective way to reduce seabird bycatch as it puts the onus on fishing masters to keep seabird bycatch below a certain limit. During my time on fishing vessels, I learnt that many of the crew and fishing masters had a good understanding of seabird bycatch and how to reduce it, and thus with the incentive to reduce seabird bycatch, fishing masters will ensure it is kept to a minimum, where possible. However due to these bycatch limitations

there are huge incentives to ensure seabird bycatch is not observed or under-reported. To ensure that seabird bycatch goes unobserved, crew could distract the observer during periods of high seabird bycatch, cut lines with hooked birds, or remove birds before they are noted by the observer. Additionally observers could be threatened or bribed to ensure that seabird bycatch is kept to below these limits.

7.2.3 Bird-scaring lines

Bird-scaring lines proved effective at reducing seabird bycatch on Asian vessels, but had little impact on seabird bycatch by South African-flagged vessels. I suspect bird-scaring lines were less effective when used in the South Africa fleet as they were probably used selectively, mostly in areas or at times of high seabird abundance, thus increasing the risk of seabird bycatch on sets when bird-scaring lines were deployed. A problem with analysing the effectiveness of bird-scaring lines is that their design varies substantially between fleets and between vessels within fleets, and thus simply recording the use/non-use of bird-scaring lines is often too crude to be informative. The number of bird-scaring lines deployed, height deployed, length of streamers, distance between streamers, number of streamers, aerial extent when in operation and length of the in-water section of a bird-scaring line all play a role in determining the effectiveness of bird-scaring lines (Melvin et al. 2004; Sato et al. 2013). Bird-scaring lines also deteriorate easily and thus it is important that the lines are regularly maintained to ensure they remain at their most effective. South African fishing masters regularly complained that bird-scaring lines became entangled in the longlines and preferred not to use them, however fishing masters of the larger Asian vessels did not appear to mind using them, and only avoided using them during strong crosswinds, when they were more likely to become entangled. My thesis did not investigate the design of bird-scaring lines but, it is imperative, in a fishery attracting many deep-diving seabird species, that compliance with this proven bycatch mitigation measure is improved upon, particularly within the South African fleet which rarely flew bird-scaring lines (6% of sets), compared to the Asian fleet (97% of sets). To further improve the effectiveness of bird-scaring lines, vessel speeds should be reduced during line setting, with weights applied to branchlines to increase line sink rates (Fig. 7.1).

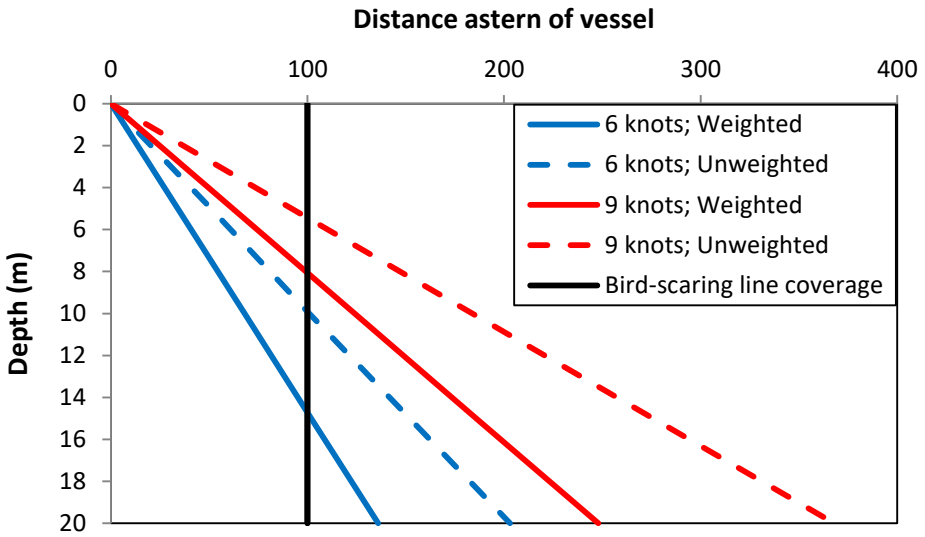


Fig. 7.1. Bird-scaring lines can protect hooks to greater depths when used in conjunction with reduced vessel setting speeds and branchline weighting. This example shows the depth to which the hooks sink by the time they leave the protection of a bird-scaring line, achieving 100 m aerial coverage behind the vessel.

7.2.4 Branchline weighting

My research in Chapter 6 shows that branchline weighting can reduce seabird bycatch without compromising crew safety or impacting fish catches or operational efficiency. South African permit conditions require that South African vessels apply a 60 g weight within 2 m of the hook in all fishing operations, while Asian vessels only need to ensure line weighting is applied when fishing outside of the South African EEZ during daylight setting. My results indicate that by using sliding leads, line weighting can be introduced safely into the Asian fleet. As a large percentage of the seabird bycatch off South Africa is represented by the nocturnally active white-chinned petrel, setting lines at night without the full suite of mitigation measures is likely to result in high levels of seabird bycatch. I thus recommend that line weighting be made mandatory for all sets, irrespective of time of setting. My results also show that applying 60 g of weight 1 m from the hook does not impact fish catches or operational efficiency, and thus I recommend that this weighting regime be used, with the option to use sliding leads to address safety concerns of fishers.

7.2.5 Night-time setting

Although night-time setting may be less effective at reducing the bycatch of white-chinned petrels than albatrosses (Barnes et al. 1997; Nel et al. 2002; Delord et al. 2010b), it has proven effective at reducing the bycatch of many other seabird species (Melvin et al. 2013). Permit conditions currently allow South African longline vessels to set longlines during daylight hours, provided their branchlines are weighted adequately. One of the significant findings of my thesis is that South African vessels are killing birds at higher rates than Asian vessels and thus I recommend the use of additional, proven seabird bycatch measures on South African vessels, including mandatory use of bird-scaring lines and night-time setting only. Additionally, the high levels of seabird bycatch on night-time sets over full moon are cause for concern. Permit conditions state that fishing for the three days around full moon is only prohibited once the limit of 25 birds in a fishing season is exceeded. I suggest that, even before the bycatch limit of 25 birds has been reached, additional mitigation measures should be required for the three days around full moon; two bird-scaring lines should be flown during line setting, appropriate weight should be applied to branchlines, vessel speeds should be reduced during line setting and hook-shielding devices could also be used.

7.2.6 Offal management

Offal management is known to affect seabird bycatch from pelagic longline fisheries during hauling activities (Gilman et al. 2005), because scavenging seabirds are more likely to be hooked at the hauling area when attracted by discards. The applicability of offal management as a means of reducing the attractiveness of the vessel to seabirds was not investigated during my study, but I gained useful insights from my time onboard pelagic longline vessels. It is difficult to entirely prohibit offal and bait discards during hauling activities, for practical reasons (lack of adequate storage facilities, etc.), due to the large quantities of bait and offal that accumulates over the course of the haul. A more realistic option would be to ensure that offal and bait are discarded on the opposite side of the vessel to where gear and fish are hauled, as suggested by ACAP (2016). This could be done by temporarily storing offal and bait in baskets/tubs, which are then emptied over the port side instead of tossing the offal off the starboard as the bait is dehooked or the fish is dressed. Although the hooking of birds during hauling does not cause high levels of mortality, it does lead to the injury of birds often of unknown severity (Brothers et al. 2010). It is also important to ensure discarding is kept to a minimum during line setting. This ordinarily

involves the discarding of cardboard packaging used to store the bait, as well as unused bait, which further attracts seabirds to the vessel during the stage when they are most vulnerable to bycatch. By retaining these items onboard during line setting, the attractiveness of the vessel to seabirds is likely to be further reduced during this critical stage.

7.2.7 Hook-shielding devices

Smart Tuna Hooks, which I tested at sea off South Africa (GB Baker unpubl. data), could potentially be incorporated into the South African longline fleet. Attachment of Smart Tuna Hooks takes a few seconds per baited hook and thus this mitigation device is only suitable for vessels which set hooks at a slow rate (c. 15 second intervals). Vessels from the Asian fleets set hooks at a faster rate (c. 6 second intervals), potentially preventing the application of Smart Tuna Hook caps. Concerns with the use of the Smart Tuna Hooks include lack of storage space to store the caps before setting and environmental concerns of discarding large amounts of metal into the ocean. An alternative to using Smart Tuna Hooks on every fishing set, would be to only use them when high bycatch levels are anticipated, such as around full moon. This latter approach would mean that storage space would be less of a concern, and significantly smaller amounts of metal would be discarded into the ocean.

7.3 Impact on seabird species

Estimates of the numbers of seabirds killed by the South African pelagic longline fishery have decreased significantly from the previous summary period, 1998–2005 (Petersen et al. 2009b; Table 7.1). Petersen et al. (2009b) raised the concern that with increasing numbers of fishing permits issued, as many as 8000 birds could be killed per year. Encouragingly, bycatch levels (both recorded and extrapolated) have not reached this magnitude, averaging c. 450 birds per year between 2006 and 2013, and with even fewer birds caught in the last four years of the study (c. 360 birds per year). A similar decrease in seabird bycatch was recorded in this same fishery between 1998 and 2005 (Petersen et al. 2009b). Changes to permit conditions and improved mitigation measure requirements are the most likely reasons for the reduced bycatch levels, although changes in seabird susceptibility cannot be excluded (Barbraud et al. 2013). The current bycatch levels by the pelagic longline fishery off South Africa are unlikely to have significant impacts on any of the species regularly recorded as bycatch (Chapter 2), but the cumulative

impacts from bycatch with other fisheries and other threats (introduced predators at breeding colonies, effects of climate change, etc.) could together have significant impacts for some populations, so there is an urgent need to reduce bycatch in all fisheries. Despite the reduced levels of seabird bycatch from pelagic longliners off South Africa, it is important to ensure that permit conditions addressing seabird bycatch are not relaxed as these have almost certainly been pivotal in reducing seabird bycatch. Future changes in regulations governing other South African fisheries also may affect seabird bycatch from longliners, either positively or negatively. For instance, Soriano-Redondo et al. (2016) found that seabird abundance and mortality associated with longline vessels increased inversely to the number of trawlers operating in proximity to the longliners. They therefore speculated that a ban on trawler discards would likely result in increased levels of seabird bycatch from pelagic longline vessels.

It is also worth considering that seabird bycatch rates may appear to have been reduced to sustainable levels by seabird-friendly fishing practices when in fact the lower bycatch rates are due to collapses in seabird populations (Tuck 2011). If this is in fact the case, one would expect a reduction in the number of seabirds attending fishing vessels, however for many fleets seabird attendance data are unavailable. Further, Barbraud et al. (2013) proposed that bycatch levels could have also been affected by the removal of a large proportion of individuals that are attracted to fishing vessels which had important phenotypic and population consequences.

Below I investigate the individual impact of the South African pelagic longline fishery on the most commonly recorded bycatch species. However, I exclude the cumulative effects from other fisheries from different regions of the world. Due to the highly migratory lifestyles of almost all procellariiform seabirds, most species will encounter many different fisheries in different fishing regions of the world throughout their lifetime. Other major fisheries such as trawl and demersal longline fisheries as well as pelagic longline fisheries operating in distant waters also have significant impacts on the species (and specific populations) that regularly occur in South African waters. Therefore the population level impacts discussed here should be considered as conservative, when the wider impacts from different fisheries/regions are taken into account.

Table 7.1. Comparison of seabird bycatch from the South African pelagic longline fishery (both foreign and local fleets) for two successive 8-year periods: 1998–2005 (A; Petersen et al. 2009b) and 2006–2013 (B; present study).

Species	**Bycatch Rate (birds per 1000 hooks)**		**Composition (%)**		**Annual bycatch**		**Total bycatch**	
	A	B	A	B	A	B	A	B
Shy-type albatrosses	0.09	0.022	24%	21%	600	97	4768	774
Black-browed albatross	0.02	0.007	4%	7%	125	19	988	151
Yellow-nosed albatrosses	0.01	0.014	2%	4%	85	25	679	198
White-chinned petrel	0.25	0.075	69%	66%	1650	227	13 185	1814
Cape gannet	-	0.005	-	2%	3*	9	-	73
All species	0.44	0.132	-	-	2890	451	23119	3604

*No Cape gannet carcasses were returned to port for autopsy by Petersen et al (2009b), however Ryan et al. (2002) reported carcasses returned to port.

7.3.1 White-chinned petrel

My thesis has focused a great deal on the foraging ecology of the white-chinned petrel, the most commonly recorded bycatch species in Southern Hemisphere pelagic longline fisheries, which is listed by the IUCN as Vulnerable (IUCN 2016). TDRs show that white-chinned petrels are capable of diving deeper than the previously recorded maximum dive depth and regularly dive deeper than 10 m (Chapter 5). Thus to ensure baited hooks are protected by bird-scaring lines to a depth of 10 m, vessel setting speeds should be reduced, with adequate line weighting used to increase hook sink rates. Due to their deep-diving capabilities and large accumulations associated with pelagic longline vessels off South Africa (Chapter 3), they are the most commonly recorded bycatch species in the region.

Analysis of white-chinned petrel movement data (Chapter 4) has shown that birds breeding on PEI make limited east-west movements, remaining between southern Africa and Antarctica, while birds from Crozet Island (Weimerskirch et al. 1999) and Kerguelen Island (Péron et al. 2010) also regularly enter South African waters to forage. Molecular studies of birds caught on South African longlines confirm that no birds were from the subspecies *P. a. steadi* that breed on New Zealand's sub-Antarctic

islands (Techow et al. 2016). As tracked white-chinned petrels (all adults) from South Georgia remained year-round near South America (Phillips et al. 2006), it appears all birds killed by pelagic longline vessels off South Africa are likely to be of the subspecies *P. a. aequinoctialis* from Crozet Island, Kerguelen Island or PEI. White-chinned petrel populations from South Georgia, southern Indian Ocean islands and New Zealand islands should thus be managed as separate stocks. Encouragingly the annual mortality of c. 230 white-chinned petrels is likely to have little impact on the combined populations of the southern Indian Ocean islands which support an estimated 294 000 breeding pairs of white-chinned petrels (Barbraud et al. 2008; Barbraud et al. 2009; Ryan et al. 2012; Table 7.2). The estimates of annual mortality have reduced dramatically from the early 2000s when 1650 white-chinned petrels were estimated to be killed each year (Petersen et al. 2009b).

Table 7.2. Impact of the South African pelagic longline fishery on the most commonly recorded seabird bycatch species. Source populations were identified conservatively; populations were excluded here if there is any doubt of the population's presence in South African waters. Estimates of fully grown birds were determined by multiplying estimated breeding pairs by four, to incorporate non-breeding adults and immatures/juveniles (Gales 1998). Annual offtake refers to the % of the source population killed by the fishery.

Species	Source of South African bycatch	Combined fully grown birds	Annual bycatch	Annual offtake (%)
White-chinned petrel	Prince Edwards, Kerguelen and Crozet Islands	1 176 000[1, 2, 3]	227	< 0.001
Shy-type albatross*	New Zealand and Tasmanian islands	460 000[4, 5]	97	< 0.001
Black-browed albatross	South Georgia	224 000[6]	19	< 0.001
Atlantic yellow-nosed albatross	Tristan da Cunha	55 600[7]	4	< 0.001
Indian yellow-nosed albatross	Prince Edwards and Crozet Islands	58 000[8]	21	< 0.001
Cape gannet	South African offshore islands	580 000[9]	9	< 0.001

* White-capped albatross are suspected to breed biennially (but not confirmed; Baker et al. 2012), therefore these estimates should be considered conservative.

Sources: [1]Ryan et al. (2012), [2]Barbraud et al. (2009), [3]Barbraud et al. (2008), [4]Baker et al. (2013), 5Alderman et al. 2011, [6]IUCN (2016), [7]ACAP (2009a), [8]ACAP (2009b), [9]Crawford et al. (2007)

7.3.2 Shy-type albatrosses

Molecular studies of shy-type albatrosses (Chapter 2) killed by pelagic longline vessels off South Africa confirm the ratio as 95% *T. steadi* and 5% *T. cauta*; consistent with the findings of Baker et al. (2007) and Petersen et al. (2009b). I therefore estimate c. 92 *T. steadi* and 5 *T. cauta* currently are killed each year by the pelagic longline fishery off South Africa. These numbers represent impressive reductions in the numbers of shy-type albatross killed from the previous estimates (Petersen et al. 2009b) and, in isolation, are unlikely to have a significant impact on either species' populations (estimated to be 15 600 breeding pairs for *T. cauta* and 100 000 for *T. steadi*; Alderman et al. 2011; Baker et al. 2013; Table 7.2). In Chapter 3 I found a disparity in numbers of shy-type albatrosses attending pelagic longline vessels and numbers recorded as pelagic longline bycatch. This disparity is perhaps best explained by the dominance hierarchy of scavenging seabirds; shy-type albatrosses, the largest of the regularly occurring seabird species to attend longline vessels off South Africa, are likely to outcompete other seabird species for baited hooks, thus experiencing elevated bycatch levels (Chapter 3).

7.3.3 Black-browed albatross

Black-browed albatross was the most commonly recorded albatross species associating with pelagic longline vessels off South Africa (Chapter 3), but many more shy albatross are killed each year by pelagic longliners (Chapter 2). As described above, black-browed albatrosses are likely to be outcompeted by the larger shy albatrosses, leading to mismatches in attendance versus bycatch figures. Birds from the decreasing South Georgian population forage off South Africa (Phillips et al. 2005; Poncet et al. 2017), however the bycatch of c. 20 black-browed albatross per year by pelagic longliners off South Africa is unlikely to adversely affect the decline of this population (Table 7.2).

7.3.4 Yellow-nosed albatrosses

Both species of yellow-nosed albatross are killed by pelagic longline vessels off South Africa and are considered Endangered by the IUCN; the Atlantic yellow-nosed from the Tristan da Cunha archipelago and Indian yellow-nosed from southwestern Indian Ocean islands. The greater numbers of Indian than Atlantic yellow-nosed albatross killed by the fishery can be explained by the far greater fishing effort east of Cape Agulhas, where Indian are more abundant (Hockey et al. 2005; Chapter 3). East of Cape

Agulhas, off the Agulhas Bank (the region with the greatest fishing effort), Indian yellow-nosed were 35 times more abundant than Atlantic yellow-nosed (Chapter 3), with ratios similar around Cape Point (pers. obs.), while Atlantic yellow-nosed dominate off the west coast of South Africa (ACAP 2009a, b). Yellow-nosed albatross spp. are the smallest, and hence most agile, of the albatross regularly attending longline vessels off South Africa; thus they frequently attack baited hooks during hauling, sometimes resulting in their non-fatal hooking (pers. obs.). Although such interactions are recorded as non-fatal, the seriousness of their injuries is often not known. The relatively low numbers of birds observed killed each year by the fishery of either Indian (c. 20 birds per year) or Atlantic yellow-nosed albatross (c. 5 birds per year) is unlikely to have significant impacts on the populations of either species which number c. 58 000 and c. 56 000 fully grown individuals, respectively (ACAP 2009a, b; Table 7.2).

7.3.5 Cape gannet

Although observers reported 20 Cape gannets killed between 1998 and 2005, none of these records were confirmed by autopsies. However, Ryan et al. (2002) reported three Cape gannets returned to port for autopsy. Between 2006 and 2013 nearly four times as many Cape gannets were killed by the South African pelagic longline fishery. Cape gannets are listed as Vulnerable by the IUCN and are restricted to only six breeding islands off South Africa and Namibia (Crawford et al. 2007). Despite some of the colonies undergoing population declines (Pichegru et al. 2007), an annual bycatch of c. 10 birds in this fishery is unlikely to have impacts on their populations (Table 7.2), with this species facing a greater risk from the South African trawl fishery (Maree et al. 2014) and the Namibian pelagic longline fishery (Petersen et al. 2007)).

7.4 Recommendations for future studies

Despite the large number of studies which have expanded our knowledge and understanding of seabird bycatch from pelagic longline fisheries, there remain a few key areas which need to be further investigated. From a South African perspective, electronic monitoring has great potential to increase our understanding of seabird bycatch, and branchline weighting should further reduce seabird bycatch in both the foreign and local fleets.

The applicability of electronic monitoring and how it compares to data collected by fisheries observers needs to be investigated locally; although studies have been conducted from fleets throughout the world's fisheries, local studies will be more meaningful. Given the large differences in vessel design and fishing operations between the local and foreign fleets, it is important to conduct studies onboard vessels from both fleets. After studies trialing electronic monitoring onboard vessels off South Africa have been conducted, recommendations can be made to governing bodies as how best to introduce electronic monitoring into both fleets. By using electronic monitoring onboard vessels which would otherwise be unobserved (such as many South African vessels), mitigation measure compliance is likely to increase, which should further reduce seabird bycatch.

In my opinion, the safe application of weight to branchlines has the greatest potential to reduce seabird bycatch from pelagic longline fisheries off South Africa. My research proving the applicability of branchline weighting for use on pelagic longliners, could be further strengthened with a study testing sliding leads onboard South African pelagic longliners. Although my study was extensive (> 200 000 hooks from three vessels), line weighting trials within a different fleet will further strengthen the argument for branchline weighting. As was the case in my study, such investigations should explore the effect of line weighting on fish catches, seabird bycatch, crew safety and operational efficiency. Additional studies will be useful when approaching bodies governing fisheries regulations, such as national departments or RFMOs. South African fishing masters are more likely to accept sliding leads if they have proven successful on other vessels from within the same fleet.

Studies investigating the effect of different bait and offal discarding regimes also would be useful. Although few birds hooked during line hauling are killed directly, some may sustain serious injuries and thus it would be beneficial to reduce these interactions. Studies quantifying the numbers of birds attracted to the line hauling area during different discarding regimes would help understand the effectiveness of discard management at reducing seabird bycatch during line hauling.

7.5 Conclusions

The aim of my thesis was to provide a broader understanding of seabird bycatch from pelagic longline vessels off South Africa and to investigate the suitability of various mitigation measures to reduce seabird bycatch. My work shows that seabird bycatch off South Africa has been significantly reduced over the last decade. Improved seabird bycatch mitigation measures as well as vessel-specific seabird

bycatch limits are the most likely factors for the significant reduction in bycatch levels. Species composition of seabird bycatch is closely linked to the composition of the seabird assemblage associated with fishing vessels as well as species' foraging ecology, particularly their diving capability. A better understanding of the distribution and foraging ecology of commonly caught seabird species, such as white-chinned petrels, will aid in the design and effectiveness of seabird bycatch mitigation measures. For example, without an understanding of the diving capabilities of seabird species prone to longline bycatch, the depth to which bird-scaring lines should provide cover for baited hooks would be unknown. Differences in the composition of seabird assemblages associating with fishing vessels, along with differences in fishing operations between worldwide pelagic longline fleets means the effectiveness of mitigation measures will vary substantially across these fleets. Therefore the trialing of mitigation measures across the different fleets is vital.

Despite reductions in the magnitude of seabird bycatch from many pelagic longline fleets from around the world, bycatch levels remain unacceptably high for other fleets. For those fleets in which seabird bycatch levels have been stabilised, it is important that regulations controlling seabird bycatch are not only retained, but improved upon, so as to further benefit conservationists as well as fishers. Human population growth, climate change and other anthropogenic influences are all likely to have unknown but largely adverse impacts on many of the oceans' resources. Seabird populations are likely to be impacted too, which in turn could affect seabird-fisheries interactions and thus seabird bycatch mortality. Seabird conservationists, working together with fishers, have made significant progress in reducing global seabird bycatch levels, yet there still remains much knowledge to be gained and action to be taken to ensure bycatch of this charismatic and vulnerable group of birds is kept to sustainable levels.

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